

In order for a body of water to be considered boatable, fishable or swimmable, it must satisfy the minimum numeric criteria consistent with that use for all modeled parameters.¹⁵ These minimum conditions are the same for all geographic areas (see Appendix 4-C).

Based on the framework described above, NWPCAM classifies each segment of each modeled river or stream as swimmable, fishable, boatable, or non-supportive of any of these uses. The model calculates the total stream-miles that support each designated use under each set of loadings conditions (i.e. baseline conditions or conditions following implementation of the revised CAFO regulations).

4.6.1.2 Carson and Mitchell Study

The contingent valuation survey upon which this analysis relies examined households' willingness to pay to maintain or achieve specified levels of water quality in freshwater lakes, rivers and streams throughout the United States (Carson and Mitchell, 1993).¹⁶ The survey was conducted in 1983 via in-person interviews at 61 sampling points nationwide, and employed a national probability sample based on the 1980 Census. Respondents were presented with the water quality ladder depicted in Exhibit 4-10 and asked to state how much they would be willing to pay to maintain or achieve various levels of water quality throughout the country. In eliciting responses, the survey used a payment card showing the amounts average households were currently paying in taxes or higher prices for certain publicly provided goods (e.g., national defense); respondents were then asked their willingness to pay for a given water quality change. The survey respondents were told that improvements in water quality would be paid for in higher product prices and higher taxes.

Exhibit 4-11 presents the results of the survey. These values represent "best estimates" of mean annual household willingness to pay (WTP) for the specified water quality improvement. Note that the values the exhibit reports are those originally obtained from the Carson and Mitchell survey, and are expressed in 1983 dollars. To provide benefit estimates appropriate for this analysis, EPA adjusts these values to account for inflation and changes in real income between 1983 and 2001.¹⁷

¹⁵ The criteria for each beneficial use category are based on criteria used by W.J. Vaughn to develop the original water quality ladder (see Carson and Mitchell (1993) for discussion of Vaughn's ladder). Vaughn's ladder included pH in addition to the four parameters adopted for this analysis.

¹⁶ The scope of the survey excluded the Great Lakes.

¹⁷ EPA employs the Consumer Price Index to adjust 1983 values to 2001 values. In addition, the adjustment to 2001 values takes into account the increase in real per capita disposable income over the period of interest. The adjustment for changes in real income is consistent with the survey's results, which found that respondents' willingness to pay for water quality improvements increased in almost direct proportion to household income.

Exhibit 4-11 INDIVIDUAL HOUSEHOLD WILLINGNESS TO PAY FOR WATER QUALITY IMPROVEMENTS (1983 \$)		
Water Quality Improvement	Total WTP	Incremental WTP
Swimmable: WTP to raise all sub-swimmable water quality to swimmable	\$241	\$78
Fishable: WTP to raise all sub-fishable water quality to fishable	\$163	\$70
Boatable: WTP to maintain boatable water quality	\$93	\$93
Source: Carson and Mitchell, 1993.		

4.6.1.3 Additional Considerations When Using the Ladder

Applying the willingness to pay estimates obtained from the Carson and Mitchell study to analyze the benefits of revised CAFO regulations requires consideration of how households' willingness to pay for water quality improvements is likely to vary with the extent and location of the resources affected. All else equal, people are likely to value an action that improves water quality along a ten-mile stretch of river more highly than they would value an action that improves only a one-mile stretch. Similarly, people are likely to place greater value on improving the quality of water resources that are nearer to them. This is simply because less time and expense is typically required to reach nearer resources; as a result, these resources generally provide lower cost and more frequent opportunities for recreation and enjoyment. This assumption is supported by the results of the Carson and Mitchell survey, which asked respondents to apportion their willingness to pay values between improving the quality of local waters — where local waters were defined as those in each respondent's own state — and improving the quality of non-local waters (i.e., those located out-of-state). On average, respondents allocated two-thirds of their values to achieving water quality goals in-state, and one-third to achieving those goals in the remainder of the nation.

To reflect the considerations noted above, the analysis of the benefits of the revised CAFO regulations examines water quality improvements on a state-by-state basis and separately calculates the benefits of in-state and out-of-state improvements, assuming that households will allocate two-thirds of their willingness to pay values to the improvement of in-state waters. In addition, the analysis takes into account the extent of the final rule's estimated impacts (i.e., the number of stream-miles that improve from non-supportive to boatable; non-supportive or boatable to fishable; or non-supportive, boatable or fishable to swimmable) by scaling household willingness to pay for a given improvement in the quality of the nation's waters by the proportion of total stream-miles in-state or out-of-state that are projected to make the improvement. Appendix 4-A provides a detailed summary of the calculations employed.

The water quality ladder captures the benefits of categorical changes in the type of beneficial uses supported by water bodies (i.e., improvements from one use category to another). In doing so, it reflects the principles of water quality standards where determinants of beneficial use attainment are based on water quality criteria. However, it should be emphasized that the pollutant criteria in the discrete ladder include pollutants (such as TSS and BOD) that are not typically adopted by States as numerical criteria for determining boatable, fishable, and swimmable conditions. In addition, the ladder criteria are relatively stringent (e.g., 100 mg/l TSS for boatable). Inclusion of criteria for these pollutants therefore implies lower probability of beneficial use attainment under the ladder than might be indicated by other methods for determining use attainment in the nation's waters. For example, 71 percent of assessed streams and rivers in the nation are judged to be supporting swimmable uses (National Water Quality Inventory (NWQI): 2000 Report) (EPA 841-R-02-001), yet only five percent of RF3 Lite reach segments are meeting swimmable criteria at baseline (i.e., in the absence of the CAFO final rule) using the ladder.¹⁸ Similar results are observed for the boatable amenity where the NWQI (2000) shows that 76 percent of the nation's assessed streams and rivers are supporting secondary contact recreation but only 14 percent of RF3 Lite reach segments are achieving boatable conditions under the ladder.

4.6.2 Water Quality Index Approach

A key limitation of the water quality ladder approach is that it only values changes in water quality to the extent that they lead to changes in beneficial-use attainment. As a result, the approach may overstate the benefits of relatively small changes that occur at the thresholds between beneficial use categories, while failing to capture the benefits of changes that occur within (i.e., without crossing) the thresholds. Furthermore, the use classification is determined by the worst individual water quality parameter. For example, if TSS changes to boatable but fecal coliform does not, the reach would still be classified as non-boatable. Finally, another limitation of the water quality ladder is that changes in nitrogen and phosphorus concentrations, both of which are CAFO parameters of interest with respect to eutrophication, are not directly included in use support determinations.

The water quality index approach is designed to address these concerns. Under this approach, NWPCAM calculates a score for each river reach based on six water quality parameters: BOD, DO, fecal coliform, total suspended solids, nitrate, and phosphate. Scores are assigned on a scale of 0 to 100, based on a weighting process that translates the six conventional water quality measures to a continuous, composite index. The weighting process reflects the judgments of a panel

¹⁸ Baseline results provided in *Estimation of National Economic Benefits Using the National Water Pollution Control Assessment Model to Evaluate Regulatory Options for Concentrated Animal Feeding Operations* - see docket.

of 142 water quality experts convened as part of a 1974 study by McClelland (McClelland, 1974).¹⁹ The impact of the revised CAFO regulations for a given river reach is measured as the change in the water quality index for that reach (i.e., the difference between the reach's score under baseline conditions and its score under the post-regulatory scenario).

To value changes in the water quality index, EPA relies on a willingness to pay function derived by Carson and Mitchell using their survey results. This equation specifies household willingness to pay for improved water quality as a function of the level of water quality to be achieved (as represented by the water quality index value), household income, and other attributes (i.e., household participation in water-based recreation and respondents' attitudes toward environmental protection). EPA estimates changes in index values using NWPCAM, and applies the willingness to pay function to estimate benefits. Based on this approach, EPA is able to assess the value of improvements in water quality along the continuous 0 to 100 point scale. Appendix 4-B specifies the willingness to pay function and describes its derivation. As with the water quality ladder approach, the calculation of benefits is developed by State and takes into account differences in willingness to pay for local and non-local water quality improvements (i.e., it assumes households will allocate two-thirds of their willingness to pay to improvements in in-State waters).

4.6.3 Additional Considerations When Applying the Index

An issue in applying the results of the Carson and Mitchell survey in the context of the water quality index is the treatment of water quality changes occurring below the boatable range and above the swimmable range. There are concerns that the survey's description of non-boatable conditions was exaggerated, which implies that willingness-to-pay estimates for improving water to boatable conditions may be biased upwards. In addition, the survey did not ask respondents how much they would be willing to pay for improved water quality above the swimmable level.²⁰ These issues increase the uncertainty associated with valuing water quality changes outside the boatable to swimmable range (i.e., for water quality index values below 26 or above 70). In recognition of this uncertainty, value estimates for changes in water quality within each range are presented separately.

In contrast to the water quality ladder, the water quality index approach maintains greater consistency with baseline water quality conditions (i.e., NWQI results). For example, 90 to 95 percent of RF3 Lite reaches are estimated to have composite index values greater than 25 (the boatable threshold in the Carson and Mitchell survey) under baseline conditions (see memorandum summarizing distribution in record). This result is similar to the baseline conditions specified by Carson and Mitchell (approximately 99 percent of the nation's freshwater is boatable) and better

¹⁹ EPA modified the original McClelland index to eliminate three parameters not modeled in NWPCAM (temperature, turbidity, and pH).

²⁰ However, respondents were made aware of the potential for water quality to improve beyond swimmable in the ladder (e.g., drinkable).

represents NWQI results where 76 percent of assessed rivers and streams are identified as supporting beneficial uses associated with secondary contact. Note also that the WTP function used in the index approach assumes decreasing marginal benefits with respect to water quality index values; this is consistent with consumer demand theory and implies that willingness to pay for incremental changes in water quality decreases as index values increase. Other advantages of the index approach, as noted in earlier sections, include the ability to capture benefits of (1) marginal changes in water quality without triggering changes in beneficial use; and (2) changes in other parameters of interest (i.e., nitrate, phosphate) that are not included in the ladder.

4.6.4 Estimated Benefits

Exhibits 4-12 and 4-13 summarize NWPCAM's estimates of the annual economic benefits of the revised CAFO regulations. Using the water quality ladder methodology, the annual benefits attributable to the regulation of Large CAFOs under EPA's chosen phosphorus-based standard are estimated to be \$166.2 million; in contrast, annual benefits under the nitrogen-based standard, which EPA considered but did not select, are estimated to be \$102.4 million.²¹ As Exhibit 4-12 shows, a large share of the benefits under both standards is realized in improving the condition of waters previously classified as non-boatable to boatable.

The estimates yielded by the water quality index approach are higher by roughly a factor of two. Applying this approach, the annual benefits attributable to the regulation of Large CAFOs under the phosphorus-based standard are estimated to be \$298.6 million. Under the nitrogen-based standards, the analysis yields estimated annual benefits of \$182.6 million.

The lower benefits estimated under the ladder approach are due, in part, to the likelihood that predicted changes in some parameters (e.g., TSS) are not sufficiently large to meet criteria necessary for changes in beneficial use, even in the case of boatable water. Under the index approach, benefits are not constrained by limiting parameters, and the benefits of all changes in water quality parameters are captured.

Apparent inconsistencies in the distribution of benefits between the two methods arise because many water bodies fail to meet boatable criteria under the ladder approach, yet estimated water quality index values for most of these same water bodies exceed the minimum threshold index of 25 for boatable waters. As a result, a majority of water quality changes under the ladder approach occur within the non-boatable category, while a majority of water quality changes under the continuous index approach create benefits in reaches that fall within the index range of 25 to 70. This occurs because the process for calculating the index provides opportunities for low concentrations of some pollutants to offset high concentrations of other pollutants, thereby driving

²¹ The results reported are limited to the impact of the revised standards on Large CAFOs. The change in standards will also affect pollutant loads from Medium CAFOs, but the analysis of these impacts was not available when this report was submitted for publication.

up the composite score. As a final note regarding the distribution of benefits, it is also possible that a regulation, such as the final CAFO rule, may affect specific geographic areas where non-boatable waters predominate, thus implying that a majority of benefits would be attributable to improvements from non-boatable to boatable conditions.

Exhibit 4-12 ANNUAL ECONOMIC BENEFIT OF ESTIMATED IMPROVEMENTS IN SURFACE WATER QUALITY: WATER QUALITY LADDER APPROACH* (2001 \$, millions)				
Regulatory Standard	Waters Improved to Boatable**	Waters Improved to Fishable**	Waters Improved to Swimmable**	Total Benefits
Phosphorus-Based	\$114.1	\$38.8	\$13.3	\$166.2
Nitrogen-Based	\$73.1	\$23.2	\$6.1	\$102.4
Source: <i>Estimation of National Economic Benefits Using the National Water Pollution Control Assessment Model to Evaluate Regulatory Options for Concentrated Animal Feeding Operations</i> (USEPA, 2002). * These figures account for changes in loadings from Large CAFOs only. The impact of revised standards on loadings from Medium CAFOs is not considered. ** Boatable benefits include only those benefits attributable to improvements from non-boatable to boatable. Benefits from improvements to other beneficial use categories appear in the other columns. For a reach that improved from non-boatable to fishable, for example, a portion of the benefits appear in the boatable column, while the remainder appears in the fishable column. Similarly, fishable and swimmable benefits include only those benefits attributable to improvements from boatable to fishable and from fishable to swimmable, respectively. Benefits from improvements to other use categories appear in the other columns as described above.				

<p align="center">Exhibit 4-13</p> <p align="center">ANNUAL ECONOMIC BENEFIT OF ESTIMATED IMPROVEMENTS IN SURFACE WATER QUALITY: WATER QUALITY INDEX APPROACH*</p> <p align="center">(2001 \$, millions)</p>				
Regulatory Standard	WQI < 26	26 < WQI < 70**	WQI > 70***	Total Benefits
Phosphorus-Based	\$10.1	\$241.5	\$47.0	\$298.6
Nitrogen-Based	\$7.2	\$135.3	\$40.1	\$182.6
<p>Source: <i>Estimation of National Economic Benefits Using the National Water Pollution Control Assessment Model to Evaluate Regulatory Options for Concentrated Animal Feeding Operations</i> (USEPA, 2002).</p> <p>* These figures account for changes in loadings from Large CAFOs only. The impact of revised standards on loadings from Medium CAFOs is not considered.</p> <p>** This category includes only the benefits attributable to improvements between 26 and 70. For example, for a reach that improved from 24 to 30, the portion of benefits from the increase from 24 to 26 appears in the WQI<26 category; the remainder appears in the 26<WQI<70 category.</p> <p>*** This category includes only the benefits attributable to improvements to a WQI >70. For a reach that improved from 24 to 80, for example, a portion of the benefits is allocated to each of the WQI<26, the 26<WQI<70, and the WQI>70 categories.</p>				

4.7 REFERENCES

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Appendix 4-A

NWPCAM CALCULATION OF THE ECONOMIC BENEFITS OF IMPROVED SURFACE WATER QUALITY: WATER QUALITY LADDER APPROACH

Definitions

N = national benefits of estimated improvements in water quality

S_j = total benefits of estimated improvements in water quality for residents of state "j"

$B_{(l,j)}$ = benefits of in-state improvements in water quality for residents of state "j"

$B_{(n,j)}$ = benefits of out-of-state improvements in water quality for residents of state "j"

M_j = total stream-miles in state "j"

M_n = total stream-miles outside state "j"

$M_{(x,j)}$ = stream-miles in state "j" that achieve water quality improvement "x"

$M_{(x,n)}$ = stream-miles outside state "j" that achieve water quality improvement "x"

H_j = total households in state "j"

WTP_x = average household willingness to pay for water quality improvement "x"

Calculations

$$N = \sum_j S_j$$

$$S_j = B_{(l,j)} + B_{(n,j)}$$

$$B_{(l,j)} = \sum_x (M_{(x,j)} / M_j)(H_j)(WTP_x)(2/3)$$

$$B_{(n,j)} = \sum_x (M_{(x,n)} / M_n)(H_j)(WTP_x)(1/3)$$

Appendix 4-B

NWPCAM CALCULATION OF THE ECONOMIC BENEFITS OF IMPROVED SURFACE WATER QUALITY: WATER QUALITY INDEX APPROACH

The following willingness-to-pay function is used to derive economic benefits using the water quality index approach. This equation was estimated and reported by Carson and Mitchell using responses from their survey sample.

$$\text{TOTWTP} = \exp [0.413 + 0.819 \times \log(\text{WQI}/10) + 0.959 \times \log(Y) + 0.207 \times W + 0.46 \times A] \quad (1)$$

where

TOTWTP	=	each household's total WTP (in 1983 dollars) for increasing water quality up to each of the three water quality index (WQI) values
Y	=	household income (sample average = \$33,170 in 1983 dollars)
W	=	dummy variable indicating whether the household engaged in water-based recreation in the previous year (sample average = 0.59)
A	=	dummy variable indicating whether the respondent regarded the national goal of protecting nature and controlling pollution as very important (sample average = 0.65).

To develop this equation, Carson and Mitchell used the water quality ladder to map each beneficial-use category to a corresponding index value (boatable = 25, fishable = 50, and swimmable = 70).

Equation 1 can also be used as a benefit-transfer function, to assess the value of increasing water quality along the continuous 100-point water quality index. Assuming that the sample averages for W and A are representative of the current population, the incremental value associated with increasing WQI from WQI_0 to WQI_1 can be calculated as

$$\begin{aligned} \Delta \text{TOTWTP} &= \exp[0.8341 + 0.819 \times \log(\text{WQI}_1/10) + 0.959 \times \log(Y)] \\ &\quad - \exp[0.8341 + 0.819 \times \log(\text{WQI}_0/10) + 0.959 \times \log(Y)] \end{aligned} \quad (2)$$

Y , in this case, would be selected to correspond to average (or median) household income in the year of the water quality change (expressed in 1983 dollars). The resulting value estimates can be inflated to current dollars based on the growth rate in the consumer price index (CPI) since 1983.

Note that Equation 2 estimates average household willingness to pay to increase *all* impaired waters addressed in Carson and Mitchell's study by the increment WQI_0 to WQI_1 . Additional adjustments, identical to those employed under the water quality ladder approach, are required to distinguish between values for local (i.e., in-state) and non-local water quality improvements.

Appendix 4-C

WATER QUALITY LADDER THRESHOLD CONCENTRATIONS

Beneficial Use	Biological Oxygen Demand (mg/L)	Total Suspended Solids (mg/L)	Dissolved Oxygen (% saturated)	Fecal Coliforms (MPN/100mL)
Swimmable	1.5	10	0.83	200
Fishable	3	50	0.64	1,000
Boatable	4	100	0.45	2,000

5.1 INTRODUCTION

Episodic fish kills resulting from manure runoff, spills, and other discharges from AFOs remain a serious problem in the United States. As described in Chapter 2, large releases of nutrients, pathogens, and solids from AFOs can cause sudden, extensive kill events.¹ In less dramatic cases, nutrients contained in runoff from AFOs can trigger increases in algae growth — often called algae blooms — that reduce concentrations of dissolved oxygen in water and can eventually cause fish to die.²

In addition to killing and harming fish directly, pollution from AFOs can affect other aquatic organisms that in turn harm fish. In particular, the Eastern Shore of the United States has been plagued with problems related to *Pfiesteria*, a dinoflagellate algae that, under certain circumstances, can transform into a toxin that attacks fish, breaking down their skin tissue and leaving lesions or large gaping holes that often result in death. The transformation of *Pfiesteria* to its toxic form is believed to be the result of high levels of nutrients in water (Morrison, 1997). Fish kills related to *Pfiesteria* in North Carolina's Neuse River have been blamed on waste spills and runoff from the state's booming hog industry (Leavenworth, 1996; Warrick, 1996).

This chapter examines the damages attributable to AFO-related fish kills and estimates the economic benefits that the revised CAFO standards would provide in reducing such incidents. As explained below, the analysis employs state data on historical fish kill events, combined with predicted reductions in the frequency of such events under the new regulations, to estimate the

¹ For example, in 1998, the release of manure into the West Branch of Wisconsin's Pecatonica River resulted in a complete kill of smallmouth bass, catfish, forage fish, and all but the hardiest insects in a 13-mile reach (Wisconsin DNR, 1992).

² For example, in 1996, the gradual runoff of manure into Atkins Lake, a shallow lake in Arkansas, resulted in a heavy algae bloom that depleted the lake of oxygen, killing many fish (Arkansas DEQ, 1997).

decrease that would occur in the number of fish killed annually in AFO-induced incidents. It then employs two alternate approaches to estimate the economic benefits associated with the predicted reduction in fish kill incidents. The first of these approaches values reduced fish mortality on the basis of average fish replacement costs; the second values reduced fish mortality on the basis of recreational anglers' willingness to pay for improved fishing opportunities.

5.2 ANALYTIC APPROACH

5.2.1 Data Sources and Limitations

EPA does not maintain a comprehensive database detailing the frequency or severity of fish kill events, and States are not required to report fish kills to EPA. As a result, the Agency lacks a uniform source of national information on which to rely in evaluating the potential impact of the revised CAFO standards on fish kill incidents.

Despite the lack of EPA reporting requirements, many states do record information on fish kills. For purposes of this analysis, EPA has compiled a database of fish kill events in 19 states. This database incorporates a range of information on each incident. Exhibit 5-1 lists the 19 states included in the database, and for each state indicates the years for which data were obtained, the total number of reported events, the average number of reported events annually, the estimated total number of fish killed in the events reported, and the average number of fish killed per event.³

As Exhibit 5-1 indicates, the data upon which this analysis relies are not comprehensive. The fish kill database excludes 31 states, including several, such as Oklahoma, that host a relatively large number of AFOs. The period of time for which data were obtained also varies from state to state; the information collected from some states, such as Missouri, covers nearly two decades, while that collected from others, such as West Virginia, covers only a few years. In addition, even in the states and years for which data were collected, it is likely that some fish kill events remain unreported, particularly if they occurred in remote areas.⁴ These data gaps introduce considerable uncertainty into the analysis.

³ EPA's database incorporates records on fish kills obtained from the Natural Resources Defense Council and the Izaak Walton League (Frey, Hooper, and Fredregill, 2000).

⁴ For instance, in 1995 the *Raleigh News & Observer* reported a 1991 manure spill incident in the North Carolina town of Magnolia that neither the town nor the responsible farm reported to state water quality officials (Warrick and Smith, 1995).

Exhibit 5-1					
FISH KILL EVENT DATA OBTAINED BY EPA					
State	Years	Recorded Events	Average Annual Events	Estimated Number of Fish Killed	Average Mortality per Event
Arkansas	1995-1999	43	8.6	108,174	2,516
Illinois	1987-1999	182	14.0	629,118	3,457
Indiana	1994-1999	163	27.2	4,901,290	30,069
Iowa	1981-1998	473	26.3	2,342,296	4,952
Kansas	1990-1999	157	15.7	574,519	3,659
Kentucky	1995-1998	62	15.5	202,912	3,273
Minnesota	1981-1991	263	23.9	607,910	2,311
Mississippi	1990-1998	167	18.6	3,065,565	18,357
Missouri	1980-1999	2,505	125.3	701,821	280
Montana	1994-1998	9	1.8	11,212	1,246
Nebraska	1991-1998	177	22.1	167,628	947
New Mexico	1995-1998	19	4.8	3,356	177
New York	1984-1996	234	18.0	915,159	3,911
North Carolina	1994-1998	206	41.2	1,020,903	4,956
Ohio	1995-1998	81	20.3	30,923	382
South Carolina	1995-1998	22	5.5	77,760	3,535
Texas	1990-1998	1,032	114.7	141,910,079	137,510
West Virginia	1995-1997	18	6.0	64,676	3,593
Wisconsin	1988-1998	70	6.4	171,131	2,445
Total		5,883	515.9	157,506,432	26,773

In addition to the data gaps cited above, the analysis is limited by inconsistencies in the information collected in state fish kill reports. Some states appear to have established consistent guidelines for investigating a kill, which often include reporting the number of stream miles or lake acres affected, estimating the number of fish killed, describing the exact location of the kill, identifying the source of the pollutants suspected to have caused the kill, and obtaining water quality samples for testing. Other states appear to gather information on an ad hoc basis. In addition, the data present a number of anomalies or other limitations. For example, 25 percent of the records

included in EPA's database give no estimate of the number of fish killed or provide only a qualitative description of the incident's magnitude. Another 13 percent of the records indicate that the number of fish killed in the event was zero.⁵ In addition, most reports do not indicate the type(s) of fish killed.

Despite the apparent limitations of these data, they are useful for purposes of this analysis. EPA's database is the most comprehensive source of information on fish kill events currently available, and in most instances characterizes the source of the pollutants that caused individual fish kill events. Thus, EPA can apply these data to characterize a baseline of kill events potentially attributable to pollution from AFOs.

5.2.2 Predicted Change in Fish Kills Under the Revised CAFO Regulations

To estimate the potential benefits of the revised CAFO regulations in reducing fish kill incidents, EPA's analysis must first assess the current — or baseline — number of AFO-related fish kills. It must then determine the impact of the new regulations in reducing these incidents. EPA's approach to this analysis is described below.

5.2.2.1 Baseline Scenario

The EPA database records fish kill events attributable to a wide range of pollutants, sources, causes, and effects. The classification of this information varies from state to state. For purposes of identifying AFO-related fish kills, EPA applies the following criteria:

- If the source of the pollution that caused a fish kill was identified as "animal feeding/waste operations," the event was classified as AFO-related.
- If the source of the pollution that caused a fish kill was identified as "agriculture" and additional information indicated that a "lagoon break," "manure," or "ammonia toxicity" was a factor, the event was classified as AFO-related.

⁵ This may be due to a variety of circumstances. In some cases, the report may accurately indicate an event in which contamination occurred (such as a manure spill or municipal waste release) but no fish were killed. In other cases, a record may indicate zero fish killed simply because investigators were unable to develop a count (e.g., because the number killed was too great to count, or because the investigation was conducted too late to determine the number killed).

On this basis, EPA has classified 482 of the fish kill events contained in its database as AFO-related. These incidents killed a reported total of approximately 4 million fish. Based on these data, EPA estimates that in the states evaluated, incidents attributable to pollution from AFOs kill an average of 351 thousand fish per year.⁶

5.2.2.2 Post-Regulatory Scenario

Due to time and resource constraints, EPA has not conducted a detailed analysis of the impact of the revised CAFO standards on the frequency or severity of fish kill events. It is likely, however, that the implementation of the new regulations will have a number of beneficial effects. For example, because more AFOs would be subject to regulation as CAFOs, the number of fish kill incidents caused by lagoon breaks and similar catastrophic events would likely diminish. In addition, the improvements in manure management practices required under the new regulations would likely reduce the chronic discharge of nutrients to the nation's waters, and thus reduce the number of fish killed as a result of severe eutrophication.

In lieu of more detailed modeling, EPA has attempted to develop a reasonable estimate of the impact of the revised CAFO standards on fish kills. The analysis begins with EPA's estimate of the number of fish killed annually by releases from AFOs. EPA multiplies this figure by the anticipated percentage reduction in nutrient loadings from the animal feeding operations modeled by NWPCAM (see Chapter 4).⁷ The resulting value represents an estimate of the reduction in the number of fish killed annually by releases from AFOs.

Because the relationship between nutrient loadings and fish kill events is complex, this approach provides only a rough approximation of the beneficial impacts of the revised regulations. To reflect the underlying uncertainty, the analysis employs two different scaling factors:

- the percentage reduction in phosphorus loadings; and
- the percentage reduction in nitrogen loadings.

⁶ EPA estimates the average number of fish killed annually in the 19 states of record by dividing the total number of fish killed in each state by the number of years for which data from the state are reported. EPA then sums the state averages to obtain the annual average for all 19 states.

⁷ The analysis of changes in loads is limited to the impact of the revised standards on Large CAFOs. The change in standards will also affect pollutant loads from medium CAFOs, but the analysis of these impacts was not available when the report was submitted for publication.

Exhibit 5-2 summarizes the estimated percentage reduction in nitrogen and phosphorus loadings under the revised CAFO standards. The exhibit presents results for both the phosphorus-based land application standard that EPA has incorporated into the final rule and the alternative nitrogen-based standard, which EPA considered but did not select. The values reported in each case are those estimated by NWPCAM for the full RF3 set of rivers and streams. The analysis uses these values, rather than those reported for the RF3 Lite subset, in order to reflect changes in loadings to small as well as large rivers and streams.⁸

Exhibit 5-2		
SCALING FACTORS¹		
Regulatory Standard	Percent Nitrogen Reduction²	Percent Phosphorus Reduction²
Phosphorus-Based	9.7	14.0
Nitrogen-Based	3.9	7.0
¹ These figures account for changes in loadings from Large CAFOs only. The impact of revised standards on loadings from Medium CAFOs is not considered. ² The load reductions reported are NWPCAM estimates for the full RF3 set of rivers and streams.		

Based on the methods described above, EPA estimates the anticipated reduction in fish kills under the revised standards. Exhibit 5-3 presents the results. As the exhibit shows, EPA estimates that under EPA's chosen phosphorus-based standard, the reduction in fish killed annually would range from 34 thousand to 49 thousand. Under the alternative nitrogen-based standard, the reduction in fish killed annually would range from 14 thousand to 26 thousand.

Exhibit 5-3		
ESTIMATED REDUCTION IN THE NUMBER OF FISH KILLED ANNUALLY DUE TO RELEASE OF POLLUTANTS FROM AFOS¹		
(thousands)		
Regulatory Standard	Nitrogen Reduction Scaling Factor	Phosphorus Reduction Scaling Factor
Phosphorus-Based	34	49
Nitrogen-Based	14	26
¹ These figures account for changes in loadings from Large CAFOs only. The impact of revised standards on loadings from Medium CAFOs is not considered.		

⁸ Chapter 4 provides additional detail on the RF3 and RF3 Lite datasets.

5.2.3 Valuation of Predicted Reduction in Fish Kills

The economic damages that stem from natural resource injuries like fish kills include the costs of restoring the resource to its prior state, any interim lost use values (e.g., the economic value of lost fishing days from the time the damage occurs until fish stocks are restored), and any interim lost non-use values. Estimating these values for a large number of heterogeneous fish kill events nationwide is infeasible without a significant investment of analytic resources. Determining full habitat restoration costs requires a case-by-case assessment of the nature of the injury and the restoration options available, while estimating interim lost non-use values requires the use of stated preference techniques to explore people's willingness to pay to avoid temporary depletions of fish stocks and associated damage to fish habitat. The economics literature does provide estimates of potential lost use values — e.g., willingness to pay for another day of fishing or willingness to pay for an additional fish caught — that could, theoretically, be applied to the analysis using a benefit transfer approach. Conducting such an assessment at a national level, however, requires general assumptions about a number of highly variable site-specific factors, such as the duration of the reduction in fish stocks, the effect of this reduction on recreational fishing activity in the affected areas, and the availability and characteristics of alternative fishing areas. Thus, an evaluation of interim lost use values is subject to considerable uncertainty.

In light of the difficulties cited above, this analysis employs two approaches to estimating the economic benefits of reducing the frequency of fish kills. The first of these approaches values reduced fish mortality based on one component of resource restoration costs: the replacement cost of the fish. The second approach is based on a review of case studies designed to assess the damages to recreational fishing values attributable to specific fish kill events. Additional information on each approach is provided below.

5.2.3.1 Replacement Cost Approach

EPA's first approach to valuing reduced fish mortality employs fish replacement cost estimates presented in a report developed by the American Fisheries Society (AFS, 1990). These replacement values incorporate the cost of raising fish at a hatchery, transporting them, and placing them in the water. As such, they provide a conservative estimate of the economic benefits of reducing the incidence of fish kills.⁹

The American Fisheries Society report provides replacement cost estimates for a variety of fish species and size categories. Unfortunately, the available data on fish kills do not always indicate

⁹ The analysis employs fish replacement costs as a proxy measure for valuing anticipated reductions in fish kill incidents. The approach does not presume that all fish killed would necessarily be restocked.

the species of fish affected, and generally do not report mortality by size of fish. In light of these limitations, EPA applies a general fish replacement cost estimate, derived by selecting species known to have been killed in incidents related to AFOs and averaging reported replacement costs for these species across all size classes. The resulting average replacement cost employed in the analysis equals \$1.37 per fish (2001 \$).¹⁰ To value the benefits of the revised regulations, the analysis simply multiplies this average replacement cost by the estimated reduction in the number of fish killed each year.

5.2.3.2 Recreational Use Value Approach

EPA's second approach to valuing reduced fish mortality relies on an analysis of recreational fishing studies conducted to assess the damages attributable to fish kill events (IEc, 2002). Although the scope of this analysis was limited, it identified two studies that provide useful insights into the valuation of fish kills.

- ▶ The first study, of an industrial spill to Indiana's White River, examined the impacts of the spill on populations of warmwater sportfish and characterized the likely reduction in recreational fishing effort until the fishery recovered. On this basis, the study estimated interim lost use damages that equate to approximately \$1.60 per fish killed (1999 \$).
- ▶ The second study evaluated the recreational fishing impacts associated with fish entrainment at two hydroelectric dams on the Potomac River. The study estimated the reduction in warmwater sportfish stocks caused by entrainment, and assumed a proportional impact on anglers' catch rates. The study then used available estimates of anglers' willingness to pay to catch an additional fish to translate the reduction in catch into economic losses. The results range from \$2.69 to \$3.69 per fish killed (1999 \$).

¹⁰ To adjust replacement costs to 2001 dollars, EPA applies the Gross Domestic Product deflator.

On the basis of these findings the analysis estimates recreational fishing damages of approximately \$2.50 per sportfish mortality (1999 \$).¹¹ EPA's database, however, suggests that approximately 10 percent of fish kill events do not involve sportfish. Thus, the analysis recommends the use of a weighted-average figure of \$2.25 per fish (1999 \$) to value the recreational use benefits of reducing fish kills. EPA's analysis of the revised CAFO regulations adopts this recommendation, employing an inflation-adjusted value of \$2.35 per fish (2001 \$).¹²

5.3 RESULTS

Exhibit 5-4 presents estimates of the annual benefits attributable to the reduced incidence of fish kills under EPA's phosphorus-based standard and under the nitrogen-based standard that EPA considered but did not select. As the exhibit indicates, the estimated benefits range from \$47 thousand to \$115 thousand annually under the phosphorus-based standard and from \$19 thousand to \$61 thousand annually under the nitrogen-based standard, depending upon the valuation approach and scaling factor employed.

Exhibit 5-4				
ESTIMATED ANNUAL BENEFITS ATTRIBUTED TO REDUCTION IN FISH KILLS ¹ (\$2001, thousands)				
Regulatory Standard	Valuation Method			
	Replacement Cost		Recreational Use Value	
	Nitrogen Scaling	Phosphorus Scaling	Nitrogen Scaling	Phosphorus Scaling
	Phosphorus-Based	\$47	\$67	\$80
Nitrogen-Based	\$19	\$36	\$33	\$61

¹ These figures account for changes in loadings from Large CAFOs only. The impact of revised standards on loadings from Medium CAFOs is not considered.

¹¹ The analysis notes that these figures reflect recreational fishing values for warmwater sportfish, primarily bass. Such values are higher than those for most other warmwater species (e.g., bullhead, catfish), but lower than those for coldwater species (e.g., trout).

¹² EPA applies the Gross Domestic Product deflator to adjust the base value to 2001 dollars.

5.4 LIMITATIONS AND CAVEATS

EPA's analysis of the benefits of the revised CAFO regulations in reducing fish kills is subject to numerous data gaps and uncertainties. In the face of these uncertainties, the analysis employs a number of simplifying assumptions and presents a range of results. The major limitations of the analysis are summarized below.

- The scope of the analysis is limited to 19 states. The data available from these states may not include all fish kill events, and the data on reported incidents often fail to include estimates of the number of fish killed. Therefore, EPA's baseline estimate is likely to understate the number of fish kill events and the total number of fish killed nationwide each year in incidents related to pollution from AFOs.
- EPA has not undertaken a detailed analysis of the impact of the revised regulations on the incidence of fish kills. In lieu of a detailed analysis, EPA assumes that fish kills attributable to releases of pollution from AFOs will be reduced in proportion to estimated reductions in loadings of nutrients from AFOs. The direction and magnitude of bias associated with these assumptions is unknown.
- To value estimated reductions in fish kill incidents, the analysis applies two approaches. The first, which employs an estimate of average fish replacement costs, ignores other aspects of the economic damages associated with fish kills (i.e., habitat restoration costs, interim lost use values, and interim lost non-use values). Thus, it likely understates the economic benefit of reducing fish kill incidents. The second, which is based on an estimate of recreational use values, rests on a limited number of studies that reflect highly variable case-specific factors, and thus is subject to considerable uncertainty.

In addition to these caveats, the analysis is limited to the impact of the revised CAFO standards on pollutant loadings from Large CAFOs. Excluding effects on Medium CAFOs from the analysis is a source of downward (negative) bias in our estimate of the economic benefits of the new standards.

5.5 REFERENCES

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6.1 INTRODUCTION

The National Oceanic and Atmospheric Administration (NOAA) has identified pathogen contamination of U.S. coastal waters as a leading cause of government restrictions on commercial shellfish harvesting. Among the sources of pollution that contribute to such contamination are animal feeding operations (AFOs) and runoff from agricultural lands. This chapter estimates the impact of pollution from AFOs on commercial access to shellfish growing waters, the resulting impact on commercial shellfish harvests, and the potential increase in harvests that would result under the revised standards governing the discharge of pollutants from CAFOs. It then uses available estimates of consumer demand for shellfish to calculate the economic benefits associated with the predicted increase in commercial shellfish harvests under the new rule.

6.2 ANALYTIC APPROACH

6.2.1 Data on Shellfish Harvest Restrictions Attributed to AFOs

EPA's analysis of the impact of pollution from AFOs on shellfish harvests is based on information from *The 1995 National Shellfish Register of Classified Growing Waters* (NOAA, 1997) and related databases. NOAA produces the Register, which is published every five years, in cooperation with the nation's shellfish-producing states, federal agencies such as the U.S. Food and Drug Administration (FDA), and the Interstate Shellfish Sanitation Conference (ISSC). Its purpose is to summarize the status of shellfish-growing waters under the National Shellfish Sanitation Program (NSSP), which ISSC administers. The NSSP establishes comprehensive guidelines to regulate the commercial harvesting, processing, and shipment of shellfish. These guidelines include the measurement of fecal coliform concentrations as an indicator of pollution in shellfish-growing waters. Based in large part upon these measurements, shellfish-growing areas are designated as approved, conditionally approved, restricted, conditionally restricted, prohibited, or unclassified, and subjected to appropriate harvest and processing standards. Exhibit 6-1 describes these standards for each designation.

Exhibit 6-1		
NSSP STANDARDS FOR CLASSIFIED SHELLFISH GROWING WATERS		
Classification	Description	Standard ¹
Approved Waters	Growing waters from which shellfish may be harvested for direct marketing.	MPN may not exceed 14 per 100 ml, and not more than 10 percent of the samples may exceed an MPN of 43 per 100 ml for a 5-tube decimal dilution test.
Conditionally Approved Waters	Growing waters meeting the approved classification standards under predictable conditions. These waters are open to harvest when water quality standards are met. At all other times these waters are closed.	
Restricted Waters	Growing waters from which shellfish may be harvested only if they are relayed or depurated before direct marketing. ²	MPN may not exceed 88 per 100 ml, and not more than 10 percent of the samples may exceed an MPN of 260 per 100 ml for a 5-tube decimal dilution test.
Conditionally Restricted Waters	Growing waters that do not meet the criteria for restricted waters if subjected to intermittent microbiological pollution, but may be harvested if shellfish are subjected to a suitable purification process.	
Prohibited Waters	Growing waters from which shellfish may not be harvested for marketing under any conditions.	NA
Unclassified Waters	Growing waters that are part of a state's shellfish program but are inactive (i.e., there is no harvesting) and unmonitored.	NA
Source: National Oceanic and Atmospheric Administration, <i>The 1995 National Shellfish Register of Classified Growing Waters</i> , obtained from: http://seaserver.nos.noaa.gov/projects/95register/ , 11 June 2000. Notes: ¹ MPN = fecal coliform most probable number (median or geometric mean). ² The process of relaying shellfish refers to the transfer of shellfish from restricted waters to approved waters for natural biological cleansing using the ambient environment as a treatment system, usually for a minimum of 14 days before harvest. Depuration is the process of removing impurities by placing the contaminated shellfish in clean water for a period of time.		

The 1995 Shellfish Register provides information on 21.4 million acres of estuarine and non-estuarine commercial shellfish-growing waters as of January 1, 1995. A companion CD contains a GIS-based database of the location of all 4,320 shellfish growing areas in 21 coastal states, the acreage of each growing area, and the species harvested.¹ These species are classified into 13

¹ The Shellfish Register includes data for the following states: Alabama, California, Connecticut, Delaware, Florida, Georgia, Louisiana, Massachusetts, Maryland, Maine, Mississippi, North Carolina, New Hampshire, New Jersey, New York, Oregon, Rhode Island, South Carolina,

categories of clams, four categories of oysters, six categories of mussels, and two categories of scallops. In most cases, each category represents a unique species (e.g., Blue Mussel (*Mytilus edulis*)), but in some instances a category may include two or more species (e.g., Other Mussels (*Mytilus galloprovincialis* and *Mytilus edulis*)). The types of species harvested vary geographically, with large differences between the East and West Coasts.

In addition to the data described above, the shellfish database notes for each growing area any harvest limitations imposed and the known or possible source(s) of pollutants causing any impairment. The list of pollutant sources includes both “Animal Feedlots” and “Agriculture Runoff.” Sources of impairment are further classified as actual or potential contributors. If a source is listed as an actual contributor, its significance as a cause of impairment is rated as high, medium, or low. Exhibit 6-2 shows the acreage of shellfish-growing waters that are potentially or known to be impaired by pollution from AFOs and/or agricultural runoff. As the exhibit indicates, AFOs and/or agricultural runoff are known or potential contributors to the impairment of more than 1.6 million acres of shellfish-growing waters.

Exhibit 6-2			
SHELLFISH HARVEST LIMITATIONS BY REGION			
Region	Approved Acres	Harvest-Limited Acres	Harvest-Limited Acres with Impacts from AFOs and/or Agricultural Runoff
North Atlantic (MA, ME, NH)	2,920,575	714,191	33,626
Middle Atlantic (CT, DE, MD, NJ, NY, RI, VA)	4,969,680	973,715	100,284
South Atlantic (FL, GA, NC, SC)	3,505,729	1,751,844	660,679
Gulf of Mexico (AL, LA, MS, TX)	3,238,431	3,067,730	718,828
Pacific (CA, OR, WA)	206,574	214,494	96,296
Total	14,840,989	6,721,975	1,609,713
Discrepancies between reported totals and sum of regional totals are due to rounding. Source: U.S. National Oceanic and Atmospheric Administration, <i>The 1995 National Shellfish Register of Classified Growing Waters</i> , U.S. Department of Commerce, Silver Spring, MD, August 1997.			

Texas, Virginia, and Washington.

6.2.2 Estimated Impact on Shellfish Harvests

As a causal factor in the imposition of government restrictions or prohibitions on shellfish harvesting, pollution from AFOs likely serves to reduce shellfish landings below levels that would otherwise be realized. To evaluate the potential beneficial effects of the new CAFO regulations, EPA's analysis begins by estimating the adverse impacts currently attributable to pollution from AFOs. The approach to this analysis involves the following steps.

- Step 1: characterize current, or baseline, annual shellfish landings.
- Step 2: estimate the area of shellfish-growing waters from which current landings are harvested.
- Step 3: calculate the average annual per-acre yield of shellfish from harvested waters.
- Step 4: estimate the area of shellfish-growing waters that are currently unharvested as a result of pollution from AFOs.
- Step 5: estimate the foregone harvest, i.e., the potential annual harvest of shellfish from waters that are currently unharvested as a result of pollution from AFOs.

Each of these steps is described in greater detail below.

6.2.2.1 Baseline Annual Shellfish Landings

To characterize the baseline quantity (Q_0) of shellfish harvested in each coastal state, the analysis relies on data collected by NOAA's National Marine Fisheries Service (NMFS), which reports commercial fishing harvests by state, year, and species (NMFS, 2000). NMFS maintains complete commercial harvest data on various species of clams, mussels, oysters and scallops for each state. The data consist of total pounds harvested and total ex-vessel revenues for harvested species. The data are provided as state-wide totals only and do not disaggregate harvest quantities between shellfish growing areas within each state. For the purpose of this analysis, EPA obtained shellfish harvest data by species and state for the five most recent years available: 1994 through 1998. The analysis employs the mean of the reported annual values for each species and state to characterize shellfish harvests under baseline conditions.²

² The calculation of the mean ignores years for which harvest data for a particular species are unavailable. If landings in these years were actually zero, this approach will overstate average annual landings.

6.2.2.2 Estimated Acreage of Harvested Waters

The available data do not indicate the distribution of shellfish landings from waters that the 1995 Shellfish Register identifies as approved, conditionally approved, restricted, or conditionally restricted. For purposes of this analysis, EPA assumes that baseline landings are harvested primarily from approved or conditionally approved waters. Thus, in a given state (j), the area of shellfish growing waters assumed to be harvested is determined by the following calculation:

$$\text{Acres Harvested}_{(j)} = \text{Acres Approved}_{(j)} + \text{Acres Conditionally Approved}_{(j)}$$

6.2.2.3 Average Annual Yield of Harvested Waters

To calculate the average annual yield (Y) of harvested waters for a given species (n) in a given state (j), the analysis simply divides the annual baseline harvest (Q_0) for that species and state by the acres assumed to be harvested:

$$Y_{(n,j)} = Q_{0(n,j)} / \text{Acres Harvested}_{(j)}$$

This calculation provides an estimate of the pounds of shellfish landed per year from harvested waters.

6.2.2.4 Characterization of Waters that are Unharvested due to Pollution from AFOs

The next step in the analysis is to estimate the area of shellfish-growing waters that are currently unharvested due, at least in part, to pollution from AFOs. Consistent with the approach outlined thus far, EPA assumes that waters classified in the 1995 Shellfish Register as restricted, conditionally restricted, or prohibited are essentially unharvested. Thus, in a given state (j), the area of shellfish growing waters assumed to be unharvested is determined by the following calculation:

$$\text{Acres Unharvested}_{(j)} = \text{Acres Restricted}_{(j)} + \text{Acres Conditionally Restricted}_{(j)} + \text{Acres Prohibited}_{(j)}$$

This calculation, however, includes all impaired waters. To identify areas impaired, in whole or in part, by pollution from AFOs, EPA's analysis considers two cases. Under Case 1, EPA evaluates only those shellfish-growing waters for which AFOs are specifically identified as a contributing source of impairment. Under Case 2, EPA expands the analysis to include shellfish-growing waters that the Register identifies as impaired, in whole or in part, by AFOs and/or agricultural runoff. The inclusion of Case 2 is justified by the classification of shellfish-growing waters on the basis of fecal

coliform levels. To the extent that agricultural runoff causes elevated fecal coliform counts, animal manure, potentially from AFOs, is the likely contributing factor.³

6.2.2.5 Estimated Impact of Pollution from AFOs on Commercial Shellfish Landings

To characterize the impact of pollution from AFOs on commercial shellfish landings, it is necessary to estimate the potential yield of impaired shellfish growing areas. For purposes of this analysis, EPA assumes that the average annual yield from harvested waters, as calculated above, is representative of the potential annual yield from impaired waters. Thus, the foregone harvest (Q_F) from an area of any size for a given species (n) in a given state (j) is calculated as follows:

$$Q_{F(n,j)} = Y_{(n,j)} \times \text{Acres Unharvested}_{(j)}$$

EPA calculates the foregone harvest for each of the two cases described above. Under Case 1, the calculation estimates the foregone harvest from shellfish-growing waters for which AFOs are specifically identified as a contributing source of impairment. Under Case 2, EPA expands the analysis to estimate the foregone harvest from shellfish-growing waters identified as impaired, in whole or in part, by AFOs and/or agricultural runoff.

6.2.3 Estimated Impact of the Revised Regulations on Commercial Shellfish Harvests

The next step in EPA's analysis is to estimate the impact of the new CAFO regulations on commercial shellfish harvests. To do so, EPA employs information obtained from the surface water quality modeling effort described in Chapter 4. The modeling exercise does not extend to estuaries or near-coastal waters, where most commercial shellfish-growing areas are located; however, it does consider the impact of the new regulations on fecal coliform counts in the terminal reaches of rivers and streams that flow into commercial shellfish growing areas. In lieu of more detailed modeling, this information provides a reasonable proxy for estimating the impact of the rule on water quality in shellfish growing areas.

EPA's approach to estimating the beneficial effects of the new CAFO regulations on commercial shellfish harvests assumes that the adverse impact of pollution from AFOs will be

³ In addition, NOAA staff who maintain the Register suggest that difficulty in pinpointing the source of pollution often results in classifying impacts from AFOs under the more general heading of "Agriculture Runoff." Personal communication with Jamison Higgins, NOAA, April 12, 1999.

reduced in proportion to modeled reductions in fecal coliform loadings to shellfish growing waters. The details of this approach are described below.

- First, EPA identifies all terminal reaches in each state that flow into waters supporting commercial shellfish beds. The total fecal coliform load from these waters is calculated under both baseline conditions and under the revised standards. The analysis examines fecal coliform loads under both the phosphorus-based land application standard incorporated into the final rule and the nitrogen-based alternative standard, which EPA considered but did not select.
- Next, for each state, EPA calculates the percentage reduction in fecal coliform loads predicted under the revised standards.⁴
- Third, EPA multiplies its estimates of the percentage reduction in fecal coliform counts by its previously developed estimates of the impact of pollution from AFOs and/or agricultural runoff on shellfish harvests (Q_F). This calculation was performed separately for each species and state. The result, Q_R , represents the incremental increase in harvest associated with the new CAFO standards.

Adding Q_R to baseline harvests (Q_0) yields an estimate of annual shellfish harvests following implementation of the revised CAFO regulations (Q_1). This calculation is performed for each state and species. Thus:

$$Q_{1(n,j)} = Q_{0(n,j)} + Q_{R(n,j)}$$

6.2.4 Valuation of Predicted Change in Shellfish Harvests

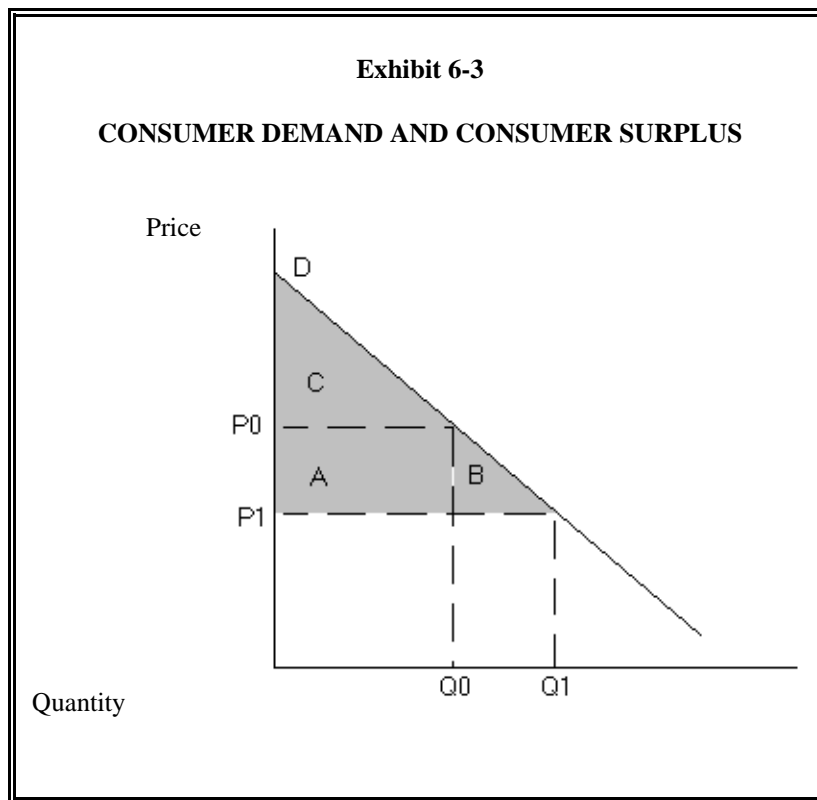
The appropriate measure of the economic benefits of an increase in commercial shellfish harvests is the welfare gain (i.e., the change in producer and consumer surplus) associated with the increased harvest. For purposes of this analysis, EPA focuses solely on changes in consumer surplus.⁵ This focus is necessary because the information required to evaluate any changes in

⁴ The analysis of changes in loads is limited to the impact of the revised standards on Large CAFOs. The change in standards will also affect fecal coliform loads from Medium CAFOs, but an analysis of these impacts was not available when this report was submitted for publication.

⁵ As discussed in Chapter 3, the concept of consumer surplus is based on the principle that some consumers benefit at current prices because they are able to purchase a good at a price that is less than the amount they are willing to pay.

producer surplus that might result from an increase in shellfish harvests (i.e., a long-run supply curve for each species harvested) is difficult to obtain. In addition, the shellfish harvesting industry is to a significant extent characterized by regulated harvest levels and unregulated harvester effort (i.e., open access fisheries).⁶ Generally accepted natural resource economics theory suggests that, in open access fisheries, overcapitalization leads to zero producer surplus. Thus, although shellfish harvesting is not entirely open access, any producer surplus in the industry is likely to be small, and any changes in producer surplus brought about by the new CAFO regulations is likely to be minor.

To calculate the change in consumer surplus associated with an increase in commercial shellfish harvests, the analysis makes use of information on consumer demand. Exhibit 6-3 illustrates a simple demand curve. The demand curve is the downward sloping solid line labeled D, and the initial quantity sold is the dashed, vertical line at Q_0 . The intersection of these two lines gives the price at which quantity Q_0 is sold. This price is marked as P_0 and represented by the dashed horizontal line. The consumer surplus for quantity Q_0 is the area below the demand curve and above the horizontal line at P_0 . That is, the consumer surplus for Q_0 is the area labeled “C” in Exhibit 6-3.



⁶ Anecdotal evidence suggests that some shellfishing areas are leased by municipalities to individual enterprises with sole rights to harvest the area. In these cases, the limits on competition could lead to positive producer surplus. The extent of this practice, however, is unclear.

The measurement of the benefits of the revised CAFO regulations relies on the assumption that a decrease in the contamination of shellfish-growing waters would increase commercial access to shellfish beds, and thus increase the quantity of shellfish supplied to consumers (i.e., an increase from Q_0 to Q_1). This in turn would result in a lower market price for shellfish (i.e., P_1). The benefit to consumers can be determined based on the old and new prices and quantities. Before the change, the area labeled “C” in Exhibit 6-3 measures consumer surplus. After the change, consumer surplus is measured by the area of A+B+C. Thus, the difference in consumer surplus between these scenarios (i.e., Area A + Area B) is the additional consumer surplus attributable to the proposed rule and the appropriate economic measure of benefits to consumers.

6.2.4.1 Characterization of Consumer Demand for Shellfish

Analysis of the changes in consumer surplus that might result from an increase in shellfish harvests requires an understanding of the effect of an increased harvest on market prices. To gather the necessary information, EPA reviewed the economics literature. This review identified a number of relevant studies: Lipton and Strand (1992), which estimates a demand equation for surf clams and ocean quahogs on the East Coast; Wessells et al. (1995), which estimates a demand equation for U.S. harvested mussels in Montreal; Cheng and Capps, Jr. (1988), which estimates demand equations for oysters and total shellfish in the U.S.; and Capps, Jr. and Lambregts (1991), which estimates demand equations for scallops and oysters in Houston, Texas. Exhibit 6-4 lists the demand elasticities obtained from each of these studies.⁷ These demand elasticities provide the means to determine the change in consumer surplus associated with changes in shellfish harvests.

Exhibit 6-4		
SHELLFISH DEMAND ELASTICITIES		
Citation	Species	Elasticity
Cheng and Capps	oysters	-1.132
Cheng and Capps	total shellfish	-0.885
Capps and Lambregts	oysters	not significant
Capps and Lambregts	scallops	-1.84
Wessells et al.	mussels	-1.98
Lipton and Strand	surf clams	-2
Lipton and Strand	ocean quahogs	-0.87

6.2.4.2 Determining the Change in Consumer Surplus Associated with Increased Harvests

⁷ The price elasticity of demand represents the percentage change in demand for a good brought about by a one percent change in its price; thus, a price elasticity of -2 implies that a one percent increase in price will result in a two percent decrease in demand.

EPA's analysis of the benefits of an increase in shellfish harvests begins by estimating prices and quantities (i.e., P_0 and Q_0) under baseline conditions, as well as the quantity of shellfish that would be harvested following the implementation of the new CAFO regulations (Q_1). Consistent with the analysis of shellfish harvests described above, Q_0 for each state and species is based on NMFS data, and specified as the mean annual harvest for the years 1994 through 1998. P_0 is calculated by dividing the total reported revenues from 1994 through 1998 for each species and state, adjusted to 2001 dollars, by the total quantity harvested.⁸ Q_1 is determined as described above, adding to Q_0 the increase in shellfish harvests estimated to occur under the new regulations (Q_R). EPA determined the value of these factors for each broad category of shellfish for which NMFS data are available: scallops, oysters, mussels, and clams. When the data allow, EPA developed separate values for quahogs, surf clams, and other clams. This approach enables the analysis to take advantage, whenever possible, of the demand equations identified for the quahog and surf clam subcategories.⁹

Once P_0 , Q_0 , and Q_1 are estimated, the appropriate price elasticities of demand are applied to determine the new price (P_1) associated with an increase in shellfish harvests. For purposes of this analysis, the percentage change in price is determined by dividing the percentage increase in the quantity of shellfish supplied in each case by the appropriate price elasticity. This percentage change is then applied to the initial price (P_0) to calculate the new price (P_1) for each species harvested.¹⁰

⁸ EPA adjusts reported revenues to 2001 dollars using the Consumer Price Index. In calculating P_0 , EPA considers only those years for which harvest and revenue data are available.

⁹ The analysis employs the Wessells et al. demand elasticity for mussels and the Capps and Lambregts demand elasticity for scallops for all states in which these species are harvested. When disaggregated data on surf clam or quahog harvests are available, the analysis relies on the demand elasticities for these species developed by Lipton and Strand; in all other instances, demand for clams is analyzed using the total shellfish price elasticity estimated by Cheng and Capps. For oysters, the analysis relies upon the demand elasticity estimated by Cheng and Capps; this value was selected because it was based on evaluation of a broader market than that considered by Capps and Lambregts.

¹⁰ Mathematically, the price elasticity of demand (ϵ) is calculated as:

$$\epsilon = \partial Q / \partial P$$

where:

$$\partial Q = (Q_1 - Q_0) / Q_0$$

$$\partial P = (P_1 - P_0) / P_0$$

therefore:

$$\partial P = \partial Q / \epsilon$$

$$P_1 = (Q_1 - Q_0)(P_0) / [(\epsilon)(Q_0)] + P_0$$

EPA employs the estimated values for P_0 , P_1 , Q_0 and Q_1 to measure the increase in consumer surplus associated with the projected increase in shellfish harvested and resulting reduction in market price under the new regulations. This calculation is conducted for every state and species category. The estimated annual benefit of the revised CAFO standards is simply the sum of the estimated increase in consumer surplus across states and species.¹¹

6.3 RESULTS

Exhibit 6-5 summarizes the estimated economic benefits associated with increased shellfish harvests under the new CAFO standards. Results are provided for both the phosphorus-based land application standard incorporated into the final rule and the nitrogen-based alternative standard, which EPA considered but did not select. The exhibit also presents two cases: Case 1, which considers beneficial impacts on shellfish growing waters that the Shellfish Register specifically identifies as impaired by pollution from AFOs; and Case 2, which expands the analysis to consider beneficial impacts on shellfish growing waters identified as impaired by pollution from AFOs and/or agricultural runoff. As the exhibit indicates, EPA's estimates of annual benefits in Case 2 are more than an order of magnitude greater than in Case 1; this range reflects the significant increase in the number and area of shellfish growing waters considered to be impaired by AFOs when runoff from agricultural land, as opposed to pollution specifically attributed to AFOs, is included in the analysis. Under EPA's chosen phosphorus-based standard, the estimate of annual benefits ranges from approximately \$0.3 million in Case 1 to \$3.4 million in Case 2. Under the alternative nitrogen-based standard, the estimates of annual benefits are lower, ranging from \$0.1 million in Case 1 to \$1.9 million in Case 2.

Exhibit 6-5		
ESTIMATED ANNUAL BENEFITS OF INCREASED COMMERCIAL SHELLFISH HARVESTS¹ (2001 \$, millions)		
Regulatory Standard	Case 1: AFOs	Case 2: AFOs and Agricultural Runoff
Phosphorus-Based	\$0.3	\$3.4
Nitrogen-Based	\$0.1	\$2.0
¹ The analysis accounts for changes in the regulation of Large CAFOs only. The impact of revised standards for Medium CAFOs is not considered.		

¹¹ The calculation of increased consumer surplus is based on a simple geometric approximation of the change in areas under the demand curve, rather than formal integration using calculus. As a result, the estimated increase in consumer surplus may be slightly overstated.

6.4 LIMITATIONS AND CAVEATS

The analysis set forth above is subject to a number of uncertainties and relies upon several simplifying assumptions. These factors may lead to a potential under- or over-estimation of the benefits of decreasing AFO-related contamination of commercial shellfish growing waters. The most significant of these limitations are described below.

- The analysis assumes that a reduction in pollution from AFOs will result in an increase in commercial shellfish harvests. While this assumption appears reasonable in light of the extent to which AFOs contribute to current restrictions or prohibitions on shellfish harvesting, the actual impact of these restrictions or prohibitions on annual shellfish landings is unknown.
- To estimate the potential impact of pollution on annual shellfish landings, the analysis calculates an average annual yield (pounds per acre) for shellfish growing waters. The calculation of this figure assumes that current harvests are obtained from waters classified as approved or conditionally approved. To the extent that this approach over- or understates the increase in annual yields that might be realized from waters currently subject to harvest restrictions or prohibitions, the analysis may either over- or understate the impact of pollution on annual shellfish landings.
- The actual contribution of AFOs to the impairment of shellfish growing waters is unclear. In light of ambiguities in the data and uncertainties associated with the impact of pollution from other sources, the analysis considers two cases to characterize the impact of pollution from AFOs on shellfish harvests. The broad range of results across the cases analyzed suggests considerable uncertainty concerning the impact of pollution from AFOs.
- Similarly, in characterizing the impact of the revised regulations, the analysis assumes that the adverse impact of pollution from AFOs (i.e., the foregone harvest) will be reduced in proportion to modeled reductions in fecal coliform loadings from rivers and streams that flow into shellfish-growing areas. While this approach may provide a reasonable approximation of the impacts of the new CAFO standards, it is less reliable than detailed modeling of pathogen concentrations in waters that support commercial shellfish beds. The direction and magnitude of any bias introduced by reliance on this approach is unclear.
- The analysis relies on estimates of the price elasticity of demand for shellfish that are not necessarily representative of current conditions or of conditions

nationwide. The direction and magnitude of any bias introduced by reliance on these estimates, however, is unclear.

Finally, the analysis is limited to the impact of the revised CAFO standards on pollutant loadings from Large CAFOs. Excluding effects on Medium CAFOs from the analysis is a source of downward (negative) bias in the estimated economic benefits of the final rule.

6.5 REFERENCES

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7.1 INTRODUCTION

CAFOs can contaminate aquifers and thus impose health risks and welfare losses on those who rely on groundwater for drinking water or other uses. Of particular concern are nitrogen and other animal waste-related contaminants (which come from manure and liquid wastes) that leach through soils and ultimately reach groundwater. Nitrogen loadings convert to elevated nitrate concentrations at household and community system wells, and elevated nitrate levels in turn pose a risk to human health.

The federal health-based National Primary Drinking Water Standard for nitrate is 10 mg/L. This Maximum Contaminant Level (MCL) applies to all Community Water Supply systems, but not to households that rely on private wells. As a result, households served by private wells are at risk of exposure to nitrate concentrations above 10 mg/L, which EPA considers unsafe for sensitive subpopulations (e.g., infants). Nitrate above concentrations of 10 mg/L can cause methemoglobinemia (“blue baby syndrome”) in bottle-fed infants (National Research Council, 1997), which causes a blue-gray skin color, irritability or lethargy, and potentially long-term developmental or neurological effects. Generally, once nitrate intake levels are reduced, symptoms abate. If the condition is untreated, however, methemoglobinemia can be fatal.¹

U.S. Census data for 1990, the most recent available for this analysis, show that approximately 13.9 million households located in counties with AFOs are served by domestic wells. A number of sources provide information on the percentage of such wells with nitrate concentrations in excess of 10 mg/L. As indicated in Exhibit 7-1, the values reported vary widely, depending on the location studied, local hydrology, and other factors. According to the nationwide USGS (1996) Retrospective Database, however, the concentration of nitrate exceeds the 10 mg/L threshold in 9.45

¹ No other health impacts are consistently attributed to elevated nitrate concentrations in drinking water. As discussed in Chapter 2, however, other health effects are suspected.

percent of domestic wells in the United States. Thus, EPA estimates that approximately 1.3 million households in counties with AFOs are served by domestic wells with nitrate concentrations above 10 mg/L.²

Exhibit 7-1			
PERCENTAGE OF DOMESTIC WELLS EXCEEDING THE MCL FOR NITRATE			
Study	Location	Type of Well	Percent Exceeding 10 mg/L
CDC, 1998	Illinois, Iowa, Kansas, Minnesota, Missouri, Nebraska, North Dakota, South Dakota, Wisconsin	Domestic	13.4%
Agriculture Canada, 1991 (as cited by Giraldez and Fox, 1995)	Ontario	Domestic farm	13%
Kross et al., 1993	Iowa	Rural	18%
Retrospective Database; USGS, 1996	National	Domestic	9.5%
Richards et al., 1996	Indiana, Kentucky, Ohio, West Virginia	Rural	3.4%
Spalding and Exner, 1993	Iowa, Kansas, Nebraska, North Carolina, Ohio, Texas	Rural	20%, 20%, 20%, 3.2%, 2.7%, 8.2%, respectively
Swistock et al., 1993	Pennsylvania	Private	9%
U.S. EPA, 1990	National	Rural domestic	2.4%
USGS, 1985	Upper Conestoga River Basin	Rural	40+%
USGS, 1998	Nemaha Natural Resources District, Nebraska	Rural	10%
Vitosh, 1985 (cited in Walker and Hoehn, 1990)	Southern Michigan	Rural	34%

² Based on analysis of the 1990 Census data, 13,871,413 households served by private wells are located in counties with AFOs. The USGS database indicates that nitrate concentrations exceed 10 mg/L in 9.45 percent of domestic wells nationwide. Applying this percentage to the figure above (13,871,413 x .0945) yields an estimate of 1,310,849 domestic wells that (1) are located in counties with AFOs and (2) exceed the MCL for nitrate.

EPA's revisions to the NPDES regulation and effluent guidelines affect the number and type of facilities subject to regulation as CAFOs, and also introduce new requirements governing the land application of manure. As a result, EPA anticipates that the revised regulations will reduce nitrate levels in household wells. In light of clear empirical evidence from the economics literature that households are willing to pay to reduce nitrate concentrations in their water supplies — especially to reduce concentrations below the MCL — the anticipated improvement in the quality of water drawn from private domestic wells represents a clear economic benefit. This chapter estimates these benefits for the final effluent guideline and final NPDES regulation.

7.2 ANALYTIC APPROACH

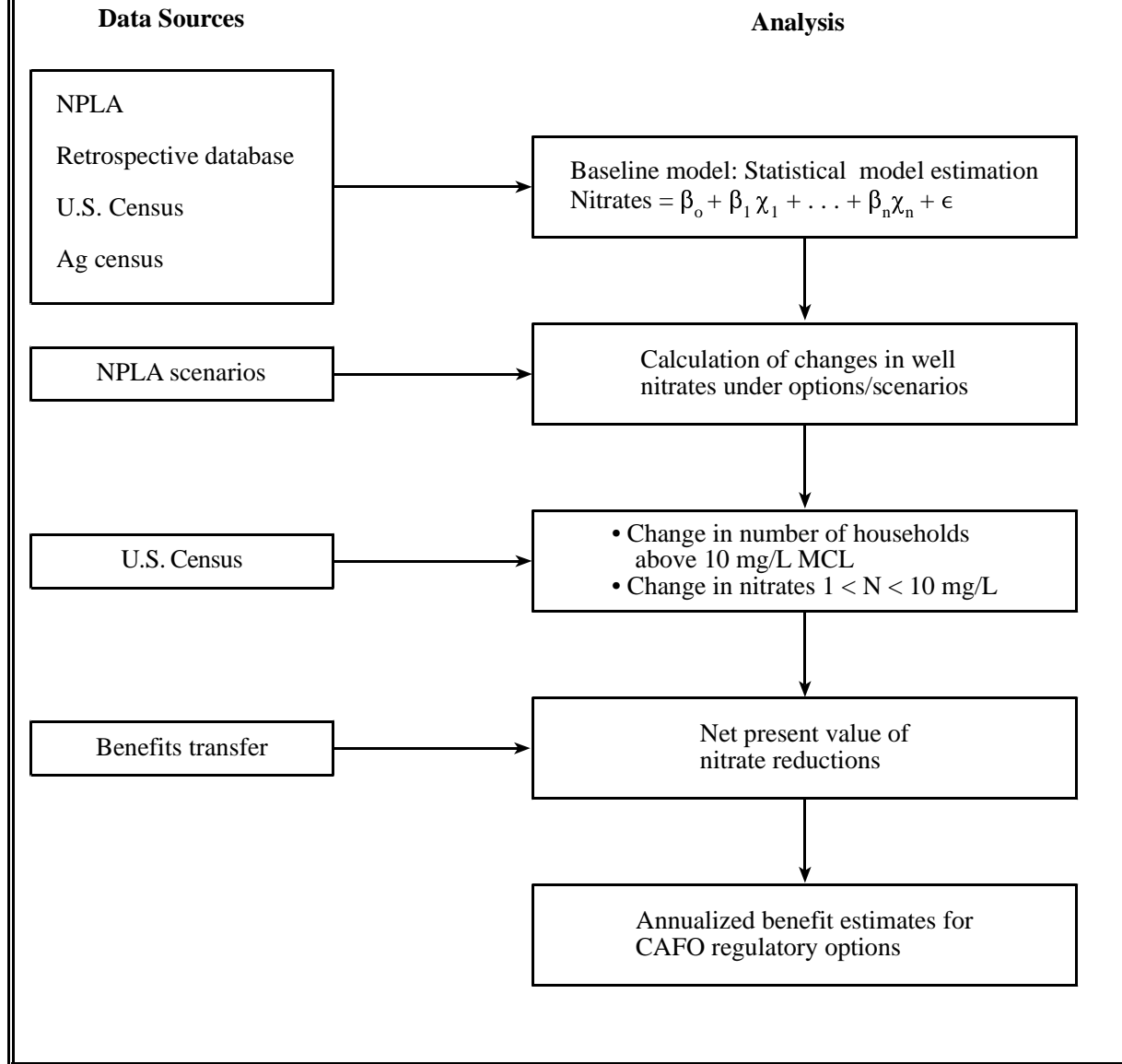
Exhibit 7-2 provides an overview of EPA's approach to estimating the benefits of well nitrate reductions. As the exhibit indicates, the analysis begins by developing a statistical model of the relationship between nitrate concentrations in private domestic wells and a number of variables found to affect nitrate levels, including nitrogen loadings from AFOs. It then applies this model, in combination with the projected change in nitrogen loadings from CAFOs, to characterize the distribution of expected changes in well nitrate concentrations. Next, the analysis applies this distribution to the number of households served by private domestic wells to calculate (1) the increase in the number of households served by wells with nitrate concentrations that are below the MCL and (2) the marginal change in nitrate concentrations for households currently served by wells with nitrate concentrations below the MCL. Finally, the analysis employs estimates of households' values for reducing well nitrate concentrations to develop a profile of the economic benefits of anticipated improvements in well water quality. Additional detail on EPA's analytic approach is provided below.

7.2.1 Relationship Between Well Nitrate Concentrations and Nitrogen Loadings

EPA's approach begins with the use of regression analysis to develop a model characterizing the empirical relationship between well nitrate concentrations and a number of variables that may affect nitrate levels, including nitrogen loadings from AFOs. The variables included in the model are based on a review of hydrogeological studies that have observed statistical relationships between groundwater nitrate concentrations and various other hydrogeological and land use factors. The following discussion describes the variables included in EPA's model and the sources of data for each variable. It also notes potentially significant variables that the model does not include. Appendix 7-A and Appendix 7-B provide additional detail on the model's development.

Exhibit 7-2

OVERVIEW OF ANALYTIC APPROACH



7.2.1.1 Included Variables and Data Sources

Although the groundwater monitoring and modeling studies that EPA reviewed covered different geographic areas and focused on varying nitrate sources (e.g., septic systems, agricultural fertilizers, animal feedlots), they often found similar significant variables. In particular, nitrogen

application or loadings rates, whether from animal wastes, private septic systems, or agricultural fertilizers, were the most consistent and significant factor affecting well nitrate levels (e.g., Burrow, 1998; CDC, 1998). EPA's model includes variables characterizing nitrogen loadings from each of these sources:

- *AFOs* — Studies that addressed the effect of animal manure production on groundwater nitrate concentrations found a positive correlation between these variables (e.g., Ritter and Chirnside, 1990; Division of Water Quality, Groundwater Section, 1998). EPA's model therefore includes a variable that characterizes nitrogen loadings from AFOs. EPA obtained data on these loadings, aggregated at the county level, from the National Pollutants Loadings Analysis (NPLA; TetraTech, 2002).
- *Septic Systems* — Several studies found that the proximity of septic systems to wells is a small, but significant, contributing factor to elevated nitrate concentrations (e.g., Carleton, 1996; Richards et al., 1996). As a proxy measure for loadings from septic systems, EPA's model includes a variable characterizing the use of private septic systems in each county. Information on septic system use was drawn from the 1990 U.S. Census.
- *Other Sources* — Several studies found that the type of crop cultivated in the vicinity of wells significantly influences well nitrate levels, reflecting variation in the crops' nutrient and water needs and suggesting that agricultural fertilizers are a significant source of nitrogen to groundwater (e.g., Swistock et al., 1993; Lichtenberg and Shapiro, 1997). EPA obtained data on nitrogen loadings associated with agricultural fertilizers from the NPLA. EPA obtained data on atmospheric deposition of nitrogen from the USGS Retrospective Database (1996).

In addition to variables characterizing nitrogen loadings, EPA's model includes the following variables describing well, soil, and land use characteristics found to significantly influence well nitrate concentrations:

- *Well Depth*: Several studies found well depth to be a significant variable, inversely correlated with well nitrate concentrations, regardless of nitrate source (e.g., Detroy, 1988; Ham et al., 1998).
- *Soil Group*: A number of studies identified at least one hydrogeological characteristic, such as aquifer composition and soil type, as a significant factor affecting well nitrate concentrations (e.g., Lichtenberg and Shapiro, 1997; Lindsey, 1997).

- *Land Use:* Agricultural land use in the vicinity of wells was found to be associated with higher groundwater nitrate in several studies (e.g., Mueller et al., 1995; Carleton, 1996).

For purposes of model development, EPA obtained data on these variables from the USGS Retrospective Database (1996).

EPA's model also includes variables that describe each well's location with respect to the five regions identified in the NPLA: Central, Mid-Atlantic, Midwest, Pacific, or South. The use of these variables helps to account for potential regional differences (e.g., differences in climate) that may affect the transfer of leached nitrogen into nitrates in groundwater, as well as geological differences that may relate to background (natural) levels of nitrate in groundwater. The states that each region encompasses are as follows:

- *Central* — AZ, CO, ID, MT, NV, NM, OK, TX, UT, WY;
- *Mid-Atlantic* — CT, DE, KY, ME, MD, MA, NH, NJ, NY, NC, OH, PA, RI, TN, VT, VA, WV;
- *Midwest* — IA, IL, IN, KS, MI, MN, MO, NE, ND, SD, WI;
- *Pacific* — AK, CA, HI, OR, WA;
- *South* — AL, AR, FL, GA, LA, MS, SC.

7.2.1.2 Omitted Variables

Because of incomplete or unreliable national data, EPA's model does not include all of the potentially significant variables identified in the literature. For example, several studies cite well construction and age as significant variables with respect to well nitrate concentrations (e.g., Spalding and Exner, 1993; Swistock et al., 1993). In general, older wells are more vulnerable to nitrate contamination because their casings are more likely to be cracked, allowing surface contaminants to enter the well. Different construction materials and methods also affect how easily nitrate or other pollutants can reach groundwater via direct contamination at the wellhead. Data on this variable, however, are often unreliable because they are generally obtained by surveying well owners and relying on their subjective assessment of when and how a well was constructed; no reliable, nationally comprehensive data on well construction are available.

Several studies also found the distance from a pollutant source to the well to be significantly correlated with well nitrate concentrations (e.g., Swistock et al., 1993; Division of Water Quality, Groundwater Section, 1998). Although spatial data for well locations are available, data on the location of animal feedlots, cropland, and septic systems are not; therefore, the model excludes this variable.

7.2.2 Modeling of Well Nitrate Concentrations

To estimate the impact of selected variables on well nitrate concentrations, EPA compiled a database of 2,985 records. Each record provides information characterizing a different well, including the observed well nitrate concentration; well location, depth, soil, and land use information; data on baseline nitrogen loadings from AFOs; and data characterizing nitrogen loadings from septic systems, agricultural fertilizer, and atmospheric deposition. EPA developed its regression model on the basis of this database.

After estimating the regression model using baseline loading information, EPA estimated expected values for well nitrate concentrations under baseline conditions and following implementation of the new CAFO regulations. Two regulatory options were analyzed: the phosphorus-based land application standard incorporated into the final rule, and a nitrogen-based application standard, which EPA considered but did not select. In each case, the calculation of expected values employed data on AFO nitrogen loadings obtained from the NPLA (Tetra Tech, 2002).³ Exhibit 7-3 summarizes the expected percentage changes in well nitrate concentrations under each regulatory standard.⁴

³ Chapter 4 provides additional information on the development of pollutant loadings estimates for both the baseline and post-regulatory scenarios. For purposes of this analysis, the characterization of post-regulatory conditions is limited to the impact of the revised standards on Large CAFOs. The impact of the revised standards on Medium CAFOs is not addressed.

⁴ Testing of EPA's model indicates that it underestimates well nitrate concentrations. As a result, comparing predicted values to observed baseline values would bias the analysis. To avoid this bias, EPA compares the well nitrate concentrations the model predicts to the values it predicts under baseline conditions. The benefits assessment is based on the resulting projected percentage changes in expected well nitrate concentrations.

Exhibit 7-3		
PERCENT REDUCTION IN PROJECTED NITRATE CONCENTRATIONS ¹		
Regulatory Standard	Projected Nitrate Concentration (mg/L)	
	Mean Percent Reduction	Median Percent Reduction
Nitrogen-based	1.8%	0.2%
Phosphorus-based	2.0%	0.2%
¹ The results reported reflect the impact of the revised standards on Large CAFOs. Impacts on Medium CAFOs are not addressed.		

7.2.3 Discrete Changes from above the MCL to below the MCL

As noted above, the most recent U.S. Census data show that approximately 13.5 million households located in counties with AFOs are served by domestic wells. The USGS Retrospective Database indicates that the concentration of nitrate in 9.45 percent of U.S. domestic wells exceeds 10 mg/L. Thus, under the baseline, EPA estimates that approximately 1.3 million households in counties with AFOs are served by domestic wells with nitrate concentrations above 10 mg/L.

To estimate the impact of the new CAFO regulations on the number of wells that would exceed the nitrate MCL, EPA applied the mean percentage reduction in nitrate concentrations predicted above to the nitrate concentration values that the USGS Retrospective Database reports. Based on the resulting values, EPA calculated the percentage reduction in the number of wells with nitrate concentrations exceeding 10 mg/L. As shown in Exhibit 7-4, it then applied these values to EPA's baseline estimate of the number of households in counties with AFOs that are served by domestic wells with nitrate concentrations above 10 mg/L. Based on this analysis, EPA estimates that the phosphorus-based regulatory standard would bring approximately 111 thousand households under the 10 mg/L nitrate threshold, while the nitrogen-based standard would have a similar effect on approximately 121 thousand households.

Exhibit 7-4		
EXPECTED REDUCTIONS IN NUMBER OF HOUSEHOLDS WITH WELL NITRATE CONCENTRATIONS ABOVE 10 mg/L ¹		
Regulatory Standard	Percentage of Wells above MCL at Baseline Expected to Achieve MCL	Reduction in Number of Households above the MCL
Nitrogen-based	9.2%	120,823
Phosphorus-based	8.5%	111,529
¹ The results reported reflect the impact of the revised standards on Large CAFOs. Impacts on Medium CAFOs are not addressed.		

7.2.4 Incremental Changes below the MCL

Households currently served by wells with nitrate concentrations below the 10 mg/L level may also benefit from marginal reductions in nitrate concentrations. For purposes of this analysis, EPA assumes that such incremental benefits would be realized only for wells with baseline nitrate concentrations between 1 and 10 mg/L; presumably, an individual would not benefit if nitrate concentrations were reduced to below background levels, which for purposes of this analysis are assumed to be 1 mg/L.⁵ Exhibit 7-5 shows EPA's estimate of the new CAFO regulations' impact on mean and median nitrate concentrations in wells with baseline values between 1 and 10 mg/L. The exhibit also indicates in each case the total expected reduction in nitrate levels, expressed in mg/L.⁶ EPA estimates that approximately 5.6 million households would benefit from these marginal reductions.

Exhibit 7-5			
MEAN AND MEDIAN REDUCTIONS IN NITRATE CONCENTRATIONS FOR WELLS WITH CONCENTRATIONS BETWEEN 1 AND 10 mg/L AT BASELINE¹			
Regulatory Standard	Mean Nitrate Reduction (mg/L)	Median Nitrate Reduction (mg/L)	Total Expected National Nitrate Reduction (mg/L)
Nitrogen-based	0.114	0.015	695,662
Phosphorus-based	0.126	0.016	768,221
¹ The results reported reflect the impact of the revised standards on Large CAFOs. Impacts on Medium CAFOs are not addressed.			

7.2.5 Valuation of Predicted Reductions in Well Nitrate Concentrations

EPA's analysis relies on a benefits transfer approach to value predicted reductions in well nitrate concentrations. EPA used three general steps to identify and apply values for benefits transfer:

⁵ EPA's analysis also ignores marginal reductions in nitrate concentrations for wells that would remain above the MCL. The Agency's review of the economics literature failed to identify studies that would provide an adequate basis for valuing such changes.

⁶ The information reported in Exhibit 7-5 pertains only to wells with baseline nitrate concentrations below the MCL. Information for wells with baseline nitrate concentrations above the MCL is not included, since the benefits associated with reducing nitrate concentrations in these wells to below the MCL are potentially captured in valuing the achievement of safe nitrate concentrations.

- (1) A literature search to identify potentially applicable primary studies.
- (2) Evaluation of the validity and reliability of the studies identified. Primary evaluation criteria included:
 - the relevance (applicability) of the commodity being valued in the original studies to the policy options being considered for CAFOs; and
 - the robustness (quality) of the original study, evaluated on multiple criteria such as sample size, response rates, significance of findings in statistical analysis, etc.
- (3) Selection and adjustment of values for application to CAFO impacts.

Appendix 7-C provides detailed information on EPA's literature search and the criteria applied to evaluate and select the studies employed in the benefits assessment.

Through its review and evaluation of the relevant literature, EPA selected three studies to provide the primary values used for the benefit transfer:

- A study by Poe and Bishop (1992), which EPA employs to value changes in well nitrate concentrations from above the MCL to below the MCL.
- A study by Crutchfield et al. (1997), which EPA employs to value marginal changes in nitrate concentrations below the MCL.
- A study by De Zoysa (1995), which EPA employs to value marginal changes in nitrate concentrations below the MCL.

The Crutchfield et al. and De Zoysa studies were rated as having similar overall quality. From each of these studies EPA identified a per milligram value for marginal changes in well nitrate concentrations; the analysis employs the average of these two values for the benefits transfer.

The discussion below briefly summarizes these studies. Additional information is provided in Exhibit 7-6.

7.2.5.1 Poe and Bishop (1992)

Poe and Bishop (1992, 1999) and Poe (1993) report on the results of a contingent valuation study conducted in rural Portage County, Wisconsin, to estimate the conditional incremental benefits

of reducing nitrate levels in household wells. The area had experienced extensive nitrate problems, and previous research suggested that 18 percent of private wells in the area exceeded the MCL. The survey comprised two stages. In the first stage, individuals were asked to submit water samples from their tap and to complete an initial questionnaire. In the second stage, individuals were provided with their nitrate test results, general information about nitrates, and a graphical depiction of their exposure levels relative to both natural levels and the MCL; they then were asked to respond to contingent valuation questions (*ex post*).

Exhibit 7-6			
SUMMARY INFORMATION ON STUDIES USED FOR BENEFITS TRANSFER			
Study Reference	Poe and Bishop	Crutchfield et al.	De Zoysa
Year of Analysis	1991	1994	1994
Place	Portage County, WI	IN, Central NE, PA, WA	Maumee River Basin, northwest Ohio
Household Water Supply/ Groundwater Use	100% on private wells	IN 73%; NE 31%; PA 47%; WA 26% nonmunicipal	Not specified
Groundwater Baseline Scenarios	An increase in the number of wells in Portage County with nitrate contamination	None given	Typical N concentrations range from 0.5 to 3 mg/L, although some are much higher
Change in Groundwater Scenario	Groundwater protection program to keep nitrate levels below EPA standards	If tap water has 50% greater N levels than EPA's MCL, how much to reduce to min. safety standards; how much to eliminate	Reduce levels to 0.5-1 mg/L
Source of Contaminants	Agricultural activities	Not specified	Agricultural fertilizer
Types of Values Estimated	Option price (use value)	Total value	Total value
Duration of Payment Vehicle	Annually, for as long as respondent lives in the county	Monthly, in perpetuity	One time
Mean Annual HH WTP in 2001 Dollars	\$536 (25% reduction in nitrates to safe level) \$629 (households with 100% probability of future contamination) — Average \$583	\$2.29 per mg/L	\$1.89 per mg/L (using 3% discount rate)

The respondents' willingness-to-pay values varied, as expected, in accordance with the results of their wells' nitrate tests and other information provided to them. Poe (1993) reports that households whose wells were considered certain at some point in the future to exceed the nitrate

MCL would be willing to pay, on average, \$629 (2001 dollars) per year for a program to keep all wells in Portage County at or below the MCL. Poe and Bishop (1999) expand on the results of the survey by developing a nonlinear valuation function that characterizes how household willingness to pay for a 25 percent reduction in well nitrate concentrations varies with the initial extent of nitrate contamination. Their analysis shows that household willingness to pay for such a program increases as baseline well nitrate concentrations increase from 2 mg/L to 14.5 mg/L, then declines to zero at a baseline concentration of approximately 22.5 mg/L. Based on their valuation function, Poe and Bishop estimate that households would be willing to pay an average of \$536 (2001 dollars) per year for a 25 percent reduction from a baseline nitrate contamination level of 14.5 mg/L. Since such a change would reduce nitrate concentrations to very near the MCL, EPA considers it representative of household willingness to pay to reduce such concentrations to safe levels. Taking the midpoint of the \$629 and \$536 values reported by Poe (1993) and Poe and Bishop (1999), respectively, EPA estimates that households whose wells exceed the nitrate MCL would be willing to pay \$583 (2001 dollars) per year to reduce nitrate concentrations to safe levels.

The reliability of these results appears to be reasonably high because the contingent valuation (CV) instrument was developed and implemented with careful attention to detail and established CV research protocol. A potential limitation is that the study is based on a relatively small sample size (480 households); however, good response rates were obtained from this sample (approximately 80 percent for the first stage and 64 percent for the *ex post* stage). The Poe and Bishop study is the only study EPA reviewed that elicited such informed *ex post* values. These value statements may be considered more reliable than others because respondents knew more about the condition of their own water supply and thus were able to make better informed decisions. Moreover, in comparison to the other studies evaluated, the value estimates from this study seemed to represent a conservative lower bound on households' values for reducing nitrates to the MCL.

7.2.5.2 Crutchfield et al. (1997)

Crutchfield et al. (1997) evaluated the potential benefits of reducing or eliminating nitrates in drinking water by estimating average willingness to pay for safer drinking water. They surveyed 800 people in rural and nonrural areas in four regions of the United States (Indiana, Nebraska, Pennsylvania, Washington) using the contingent valuation method (CVM) and posing questions in a dichotomous choice format. Respondents were specifically asked what they would be willing to pay to have the nitrate levels in their drinking water (a) reduced to "safe levels" and (b) completely eliminated. Respondents were told that this would be accomplished using a filter installed at their tap, and the cost would be included in their monthly water bill. Respondents were also asked questions regarding sociodemographic characteristics such as income, age, education, and whether they currently use treated or bottled water. Across all regions, the resulting household willingness to pay to reduce nitrates to safe levels ranged from \$45.42 per month to \$60.76 per month, with a mean of \$52.89 (1994 dollars). The willingness to pay to completely remove nitrates from drinking water ranged from \$48.26 per month to \$65.11 per month, with a mean of \$54.50 (1994 dollars). The study found two variables to be significantly related to a respondent's willingness to pay: "years

lived in ZIP code,” which was positively correlated with willingness to pay, and “age of respondent,” which was negatively correlated.

7.2.5.3 De Zoysa (1995)

De Zoysa (1995) applied the contingent valuation method to evaluate the benefits of a number of programs to enhance environmental quality in Ohio’s Maumee River basin, including a program to stabilize and reduce groundwater nitrate levels. The study solicited willingness-to-pay values from residents of both rural and urban areas in the river basin, as well as residents of one out-of-basin urban area. A portion of respondents were asked whether they would pay different amounts, via a one-time special tax, to reduce nitrate contamination from fertilizer applied to fields. Under the hypothetical scenarios, nitrate concentrations would be reduced from the current range of 0.5-3.0 mg/L to a range of 0.5-1.0 mg/L. Individuals were also asked questions regarding sociodemographic characteristics, preferences for priorities for public spending, and how they used the resource in question. Based on the lower bound of the mean values reported, the study found an average one-time household willingness to pay of \$52.78 (1994 dollars) for a 1 mg/L reduction in groundwater nitrate concentrations. The study also found that income, the level of priority placed on groundwater protection, and interest in increasing government spending on education, healthcare, and vocational training all were positively and significantly correlated with willingness to pay to improve groundwater quality.

7.2.5.4 Adjustments to the Values

EPA employs the results of the Crutchfield et al. and De Zoysa reports to estimate annual household willingness to pay to reduce well nitrate concentrations when those concentrations are already below the nitrate MCL. EPA derives the appropriate value from Crutchfield by comparing the reported monthly willingness-to-pay values for reducing nitrate concentrations from above the MCL to the MCL and from above the MCL to zero. The difference between these values is \$1.61 per month. For a change between the MCL of 10 mg/L and 0 mg/L, this represents a per mg/L monthly willingness to pay of \$0.16, or \$1.92 annually (1994 dollars). To derive a comparable annual value from De Zoysa, EPA annualizes the willingness to pay value obtained from that study – an average one-time household willingness to pay of \$52.78 (1994 dollars) for a 1 mg/L reduction in groundwater nitrate concentrations – using an annual discount rate of 3 percent. This calculation yields an estimated annual household willingness to pay for a 1 mg/L reduction in nitrate concentrations of \$1.58 (1994 dollars). EPA applied the Consumer Price Index (CPI) to convert these values to 2001 dollars.⁷ The Agency then applied the midpoint of the two values, \$2.09 per mg/L per household per year, to value changes in well nitrate concentrations between 10 mg/L and 1 mg/L. Reductions in well nitrate concentrations below 1 mg/L are not valued, since EPA assumes a natural nitrate background level of 1 mg/L.

⁷ CPI-U Series ID CUUR0000SA0, not seasonally adjusted, U.S. city average, all items.

As noted above, EPA relies on the findings of Poe and Bishop to estimate that households whose wells exceed the nitrate MCL would be willing to pay \$583 (2001 dollars) per year to reduce nitrate concentrations to safe levels. These values are expressed as willingness to pay per year as long as the individual lives in the county, and thus can be directly translated to value the benefits of the new regulations.

Exhibit 7-7 summarizes the point value estimates used for benefits transfer.

Exhibit 7-7		
WILLINGNESS-TO-PAY VALUES APPLIED TO BENEFITS TRANSFER		
Study	Value	2001\$
Poe and Bishop	Annual WTP to reduce nitrate to below 10 mg/L	\$583.00
Average of Crutchfield et al. and De Zoysa	Annual WTP per mg/L between 10 mg/L and 1 mg/L	\$2.09

7.2.5.5 Timing of Benefits

It is unlikely that changes in CAFO regulations would immediately result in the changes in well nitrate concentrations that EPA's statistical model predicts. While hydrogeological conditions and other factors may vary significantly from case to case, considerable time may pass before most wells reach the steady state nitrate concentrations the model forecasts. Therefore, it is necessary to develop a time profile of the anticipated benefits of revised CAFO standards.

EPA estimates that approximately 75 percent of affected wells would realize the full benefits of reduced nitrogen loadings within 20 years (Hall, 1996). Assuming that the number of wells achieving new steady state conditions increases linearly over time, this translates to approximately 3.7 percent of wells achieving new steady state conditions each year. At this rate, all affected wells would achieve new steady state conditions in approximately 27 years. For purposes of characterizing the benefits of reduced contamination of private wells, EPA's analysis adopts these assumptions.

7.3 RESULTS

7.3.1 Annual Benefits over Time

Exhibit 7-8 illustrates the time profile of benefits for EPA's revisions to the CAFO rule. For the phosphorus-based application standard that EPA selected, the annual benefits attributable to the new regulations on Large CAFOs increase from approximately \$2.3 million in the first year following implementation to \$66.6 million in the twenty-seventh and subsequent years. For the nitrogen-based application standard, which EPA considered but did not select, the annual benefits attributable to the new regulations on Large CAFOs increase from approximately \$2.5 million in the

first year following implementation to \$71.9 million in the twenty-seventh and subsequent years. Exhibit 7-9 summarizes the estimated annual benefits once steady state conditions are achieved under both regulatory standards. As the exhibit indicates, these benefits are estimated to be \$72 million under the nitrogen-based standard and \$67 million under the phosphorus-based standard.

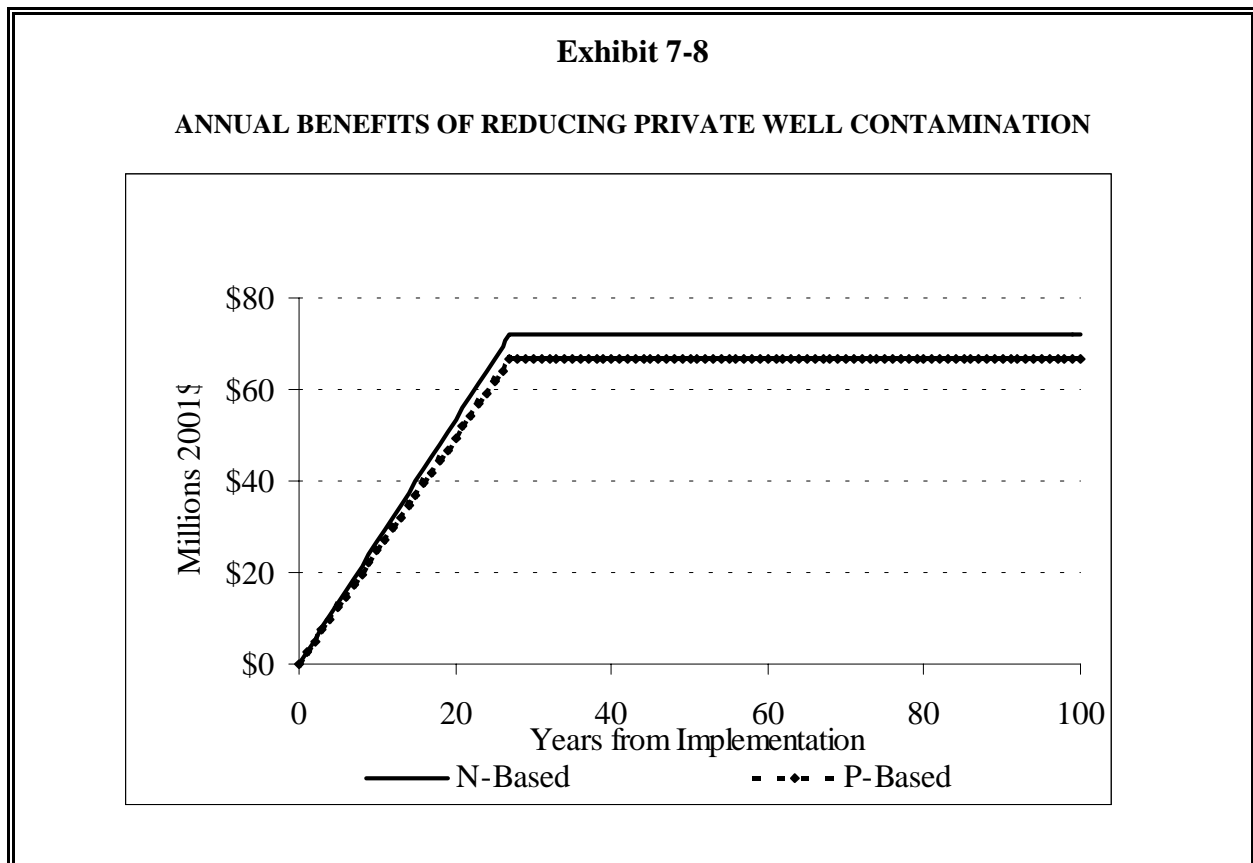


Exhibit 7-9

**ESTIMATED ANNUAL BENEFITS OF REDUCED
CONTAMINATION OF PRIVATE WELLS UNDER STEADY
STATE CONDITIONS¹**
(2001 \$, millions)

Regulatory Standard	Annual Benefits
Nitrogen-based	\$71.89
Phosphorus-based	\$66.63

¹ The results reported reflect the impact of the revised standards on Large CAFOs. Impacts on Medium CAFOs are not addressed.

7.3.2 Annualized Benefits

As discussed above, the benefits associated with reduced contamination of private wells are likely to increase for a number of years, until steady state conditions are reached. This is in contrast to the estimates of benefits developed in previous chapters, which EPA assumes will be constant over time. To report all benefits on a comparable basis, it is necessary to calculate the constant stream of benefits — the "annualized" benefits — that would yield the same present value as the uneven stream of benefits described above.

Exhibit 7-10 presents EPA's estimate of the annualized benefits associated with the reduction of nitrate concentrations in private wells under both the proposed phosphorus-based standard and the alternate nitrogen-based standard. As the exhibit indicates, the calculation of annualized benefits depends on the discount rate employed — 3, 5, or 7 percent — with lower rates yielding higher benefits.⁸ Under the phosphorus-based standard, the annualized benefits attributable to the new regulations for Large CAFOs range from approximately \$30.9 million to \$45.7 million per year. The benefits under the nitrogen-based standard range from \$33.3 million to \$49.3 million per year.

Exhibit 7-10						
ESTIMATED ANNUALIZED BENEFITS OF REDUCED PRIVATE WELL CONTAMINATION						
(2001 \$, millions)						
Regulated Entities	Nitrogen-Based Standard			Phosphorus-Based Standard		
	Discount Rate			Discount Rate		
	3 Percent	5 Percent	7 Percent	3 Percent	5 Percent	7 Percent
Large CAFOs	\$49.29	\$39.98	\$33.34	\$45.68	\$37.05	\$30.90

Under both regulatory standards, the benefits are achieved largely as a result of reducing the concentration of nitrate in private wells from above to below the 10 mg/L MCL. As discussed above, EPA estimates the value of these reductions, based on willingness-to-pay studies, to be \$583 annually (2001\$) per household. Under the nitrogen-based standard, for Large CAFOs, the total annualized value of these reductions is estimated to be \$32.7 million to \$48.3 million. Under EPA's chosen phosphorous-based standard, for Large CAFOs, the total annualized value of these reductions is estimated to be \$30.2 million to \$44.6 million. Another 5.6 million households that currently have nitrate levels in their private wells below the MCL are predicted to experience further reductions in nitrate levels because of this rule. EPA estimates a willingness-to-pay value of \$2.09 per mg/L for such reductions. For Large CAFOs, these additional reductions provide estimated annualized

⁸ Chapter 8 provides additional information on the selection of discount rates, the calculation of present values, and the calculation of annualized benefits.

benefits of \$0.7 million to \$1.0 million under the nitrogen-based standard and \$0.7 million to \$1.1 million under EPA's chosen phosphorous-based rule.

7.4 LIMITATIONS AND CAVEATS

Omissions, biases, and uncertainties are inherent in any analysis relying on several different data sources, particularly those that were not developed specifically for that analysis. Exhibit 7-11 summarizes key omissions, uncertainties, and potential biases for this analysis.

Exhibit 7-11		
OMISSIONS, BIASES, AND UNCERTAINTIES IN THE NITRATE LOADINGS ANALYSIS		
Variable	Likely Impact on Net Benefit	Comment
<i>Well, Land, and Nitrate Data</i>		
Geographic coverage	Unknown	Data availability limited the well samples used in the statistical modeling to those from 374 counties nationwide.
Well location selection	Unknown	Wells sampled in the USGS Retrospective database may not be random. Samples appear to be focused on areas with problems with high levels of agricultural activities and possibly higher nitrate levels.
Year of sample	Unknown	Samples taken over 23 years. Land use and other factors influencing nitrate concentrations in the vicinity of the well may have changed over time.
Nitrate loadings from AFOs with 0-300 AU	Positive	Data for the smallest AFOs were not included in this analysis because they will not be affected by the revised regulations. This may subsequently underestimate total loadings, resulting in an overestimate of the impact of nitrogen loadings on well nitrate concentrations.
Percent of wells above 10 mg/L	Unknown	Based on the USGS Retrospective Database, EPA assumes that 9.45 percent of wells currently exceed the MCL. If the true national percent is lower (higher), EPA's analysis overstates (understates) benefits.
Sampling methods	Unknown	Data set compiled from data collected by independent state programs, whose individual methods for measuring nitrate may differ.
<i>Model Variables</i>		
Well construction and age	Unknown	No reliable data available nationally.
Spatial data	Unknown	No national data available on the distance from well to pollutant source.
<i>Benefit Calculations</i>		
Per household value for reducing well nitrates to the MCL	Negative	The Poe and Bishop values generally appear to be a lower bound estimate of households' WTP for reducing nitrates to the MCL.
Years until wells achieve steady state.	Negative	The analysis assumes a linear path over 27 years until reduced nitrogen loadings would result in most wells achieving reduced nitrate concentrations. A large portion of wells (especially shallower wells) may achieve this much faster.
Values for marginal reductions below the MCL	Positive	If most of the benefits from reductions in nitrate concentrations below the MCL are related to a threshold effect or removing all human induced nitrates, then the assumption that benefits increase linearly with reductions in nitrate concentrations from 10 mg/L to 1 mg/L will overstate the benefits of marginal reductions.

Exhibit 7-11

OMISSIONS, BIASES, AND UNCERTAINTIES IN THE NITRATE LOADINGS ANALYSIS

Variable	Likely Impact on Net Benefit	Comment
Baseline characterization	Negative	Baseline well concentrations are based on observed levels that are in some cases more than 20 years old. These reflect AFO loadings from past decades that most likely understate current loadings and, hence, underestimate anticipated well concentrations absent regulations.
Exclusion of values for reduced nitrate concentrations in wells that would remain above the MCL after the implementation of new regulations	Negative	Reductions in nitrate concentrations in wells that would remain above the MCL after the implementation of new regulations are not valued. The Agency's review of the economics literature failed to identify studies that would provide an adequate basis for valuing such changes.
Exclusion of values for marginal reductions in nitrate concentrations below the MCL, for wells with nitrate concentrations above the MCL at baseline and below the MCL after implementation of new regulations	Negative	The benefits of marginal changes in nitrate concentrations between 10 mg/L to 1 mg/L for wells with nitrate levels above the MCL at baseline and below the MCL after implementation of new regulations are not calculated. These benefits are potentially captured in valuing the achievement of safe nitrate concentrations.
Percent change in well nitrate levels.	Positive	Poe and Bishop values are based on a 25% reduction from current levels. Modeled changes in nitrate levels for wells crossing from above to below the MCL are considerably less than 25% on average. To the extent that the value from moving from above to below the MCL is for the absolute change in nitrate levels rather than from the threshold effect, the WTP estimates used from Poe and Bishop will overstate values.

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Appendix 7-A

MODEL VARIABLES

EPA's statistical analysis of the relationship between nitrogen loadings and well nitrate concentrations is based on the following linear model:

$$\text{Nitrate (mg/L)} = \beta_0 + \beta_1 \text{ Ag Dummy} + \beta_2 \text{ Soil Group} + \beta_3 \text{ Well Depth} + \beta_4 \text{ Septic Ratio} \\ + \beta_5 \text{ Atmospheric Nitrogen} + \beta_6 \text{ Loadings Ratio} + \beta_7 \text{ Regional Dummies} + \varepsilon_i,$$

where nitrate concentration (mg/L) is the dependent variable.

The variables used to explain nitrate concentrations in well water (i.e., the model's independent variables) can be classified into two groups: well and land characteristics, and nitrogen inputs. Definitions of these variables are provided below. Unless otherwise noted, EPA obtained the data used in developing the model from the USGS Retrospective Database.

Well and Land Characteristics

Ag Dummy: This variable describes the predominant land use at the well's location (1 for agricultural land; 0 otherwise). Other land uses identified in the database include woods, range, urban, and other.

Soil Group: The soil group variable is an index that integrates several factors — including runoff potential, permeability, depth to water table, depth to an impervious layer, water capacity, and shrink-swell potential — to characterize hydrological conditions in the vicinity of the well. Values range from a minimum of 1 to a maximum of 4.

Well Depth: The well depths reported in the USGS database range from 1 foot to 5,310 feet. For observations used in the regression analysis, the maximum well depth is 1,996 feet.

Nitrogen Inputs

Loadings Ratios and Analysis of New Regulations: The loadings ratio is the sum of three variables measuring pounds of leached nitrogen per acre in each county from three different sources: CAFOs, the application of manure from CAFOs, and commercial fertilizers (because of the correlation between these nitrogen input measures, EPA was not able to estimate their parameters separately). The loadings ratio is a unique value for each county. It is calculated by dividing estimated leached nitrate loadings for the county (pounds per year) by the county's total area (acres). The analysis employs baseline loadings data to estimate the coefficients for the independent

variables. It applies these coefficients, combined with loadings data representative of post-regulatory conditions, to estimate changes in well nitrate concentrations under the new regulations.

Septic Ratio: The septic ratio is a proxy measure of potential nitrogen loadings from septic systems. The analysis develops a unique value for each county. This value is calculated by dividing the number of housing units in the county that use septic systems by the county's total area (acres). EPA obtained data on septic system use from the 1990 U.S. Census.

Atmospheric Nitrogen: The atmospheric nitrogen variable accounts for nitrogen loadings from atmospheric deposition. Values for this variable are reported in pounds per acre per year.

Regional Dummies: The regional dummy variables describe the well's location with respect to the five regions identified in the NPLA: Central, Mid-Atlantic, Midwest, Pacific, or South. The variable is assigned a value of 1 for the region in which the well is located, and a value of zero for all other regions. The use of these variables helps to account for potential regional differences (e.g., differences in climate) that may affect the transfer of leached nitrogen into nitrates in groundwater, as well as geological differences that may relate to background (natural) levels of nitrate in groundwater.

Summary Statistics

Exhibit 7A-1 reports summary statistics on the variables used in the analysis.

Exhibit 7A-1					
SUMMARY STATISTICS					
Variable	N	Mean	Standard Deviation	Minimum	Maximum
Nitrate Concentrations	2985	3.569668	6.514109	0.05	84.3
Loadings Ratio	2985	2.023526	4.156983	0.001196	18.950392
Atmospheric Nitrogen	2985	5.071787	1.865252	0.5375	8.921875
Well Depth	2985	170.0693	136.1121	1	1996
Soil Group	2985	2.422781	0.655885	1	4
Septic Ratio	2985	0.028794	0.027698	0.000217	0.151336
Ag Dummy	2985	0.776214	0.41685	0	1
Central Region Dummy	2985	0.064657	0.24596	0	1
Mid-Atlantic Region Dummy	2985	0.3933	0.488564	0	1
Pacific Region Dummy	2985	0.123953	0.329583	0	1
South Region Dummy	2985	0.070687	0.256344	0	1

Appendix 7-B

THE GAMMA MODEL

The analysis uses a gamma model to fit the right skew of observed values for well nitrate concentrations as well as the nonnegative constraint on the dependent variable. Visual inspection of the nitrate concentration distribution suggests a gamma distribution with density function:

$$f(y) = \frac{\theta^\alpha}{\Gamma(\alpha)} \exp(-\theta y) y^{\alpha-1}$$

For this distribution, the expected value of y_i is:

$$E(y_i) = \alpha/\theta_i = \alpha \exp(\beta\chi_i)$$

The use of the gamma distribution instead of the more commonly employed exponential distribution is appropriate because α is assumed to equal 1 in the exponential distribution, but was estimated to be significantly different than 1 in EPA's empirical work. The gamma distribution also offers the advantages of making the density function more flexible and giving more curvature to the distribution. The likelihood function is:

$$\log L(y_i|x_i; \alpha, \beta) = \sum_i [\alpha \log \theta_i - \log \Gamma(\alpha) - \theta_i y_i + (\alpha - 1) \log(y_i)]$$

Exhibit 7B-1 provides statistical results from the gamma model. All coefficients are of the expected sign. The coefficient for the loadings ratio variable is significant and positive, indicating that an increase in nitrogen loadings leads to increased well nitrate concentrations.

Exhibit 7B-1**GAMMA REGRESSION RESULTS**

Variable	Parameter Estimate	Standard Error	Asymptotic T-Statistic	Significance
Intercept	2.2013	0.1939	11.352	0.000
Loadings Ratio	0.0456	0.0070	6.543	0.000
Atmospheric Nitrogen	0.0315	0.0275	1.144	0.2527
Well Depth*	-0.1705	0.0124	-13.782	0.000
Soil Group	-0.3844	0.0444	-8.660	0.000
Septic Ratio	1.6179	1.7278	0.936	0.3491
Ag Dummy	0.6856	0.0643	10.663	0.000
Central Region Dummy	-0.0757	0.1596	-0.475	0.6350
Mid-Atlantic Region Dummy	-0.1654	0.0978	-1.691	0.0908
Pacific Region Dummy	0.8117	0.1173	6.918	0.000
South Region Dummy	-0.9073	0.1265	-7.170	0.000
Alpha	0.4967	0.0098	50.639	0.000
Mean log-likelihood = -1.85646				
N = 2,985				
*In the model, well depth is scaled to units of hundreds of feet.				

Appendix 7-C

LITERATURE SEARCH AND EVALUATION

Literature Search

The objective of EPA's literature search was to identify prior studies that had developed or elicited values for changes in groundwater quality, focusing in particular on values for reduced nitrates. The search drew in part on two databases: the Colorado Association of Research Libraries (CARL), which includes the holdings of several university libraries in Colorado and the West; and the Environmental Valuation Resource Inventory (EVRI), a database compiled by Environment Canada that includes empirical studies on the economic value of environmental benefits and human health effects. In addition, EPA solicited suggestions for studies pertaining to groundwater valuation and nitrate contamination through the ResEcon listserver, which reaches a network of approximately 700 academics, professionals, and other individuals with interests in natural resource and environmental economics. Through this extensive search and additional review of selected bibliographies, EPA identified 11 potentially relevant studies. Since most households' values for reducing nitrates in private domestic wells are primarily nonmarket values, most of the identified studies involve stated preference value elicitation (e.g., contingent valuation).

Evaluating Studies for Benefits Transfer

The economics literature suggests several criteria in evaluating primary studies for undertaking benefits transfer. Desvousges et al. (1992) develop five criteria to guide the selection of studies for application to a surface water quality issue: that the studies to be transferred (1) be based on adequate data, sound economic method, and correct empirical technique (i.e., "pass scientific muster"); (2) evaluate a change in water quality similar to that expected at the policy site; (3) contain regression results that describe willingness to pay as a function of socioeconomic characteristics; (4) have a study site that is similar to the policy site (in terms of site characteristics and populations); and (5) have a study site with a similar market as the policy site. NOAA condenses the five Desvousges criteria into three considerations: (1) comparability of the users and of the resources and/or services being valued and the changes resulting from the discharge of concern; (2) comparability of the change in quality or quantity of resources and/or services; and (3) the quality of the studies being used for transfer [59 FR 1183]. In a general sense, items (2), (4), and (5) of Desvousges et al. and items (1) and (2) of NOAA are concerned with the *applicability* of an original study to a policy site. Items (1) and (3) of Desvousges et al. and item (3) of NOAA are concerned with the *quality* of the original study.

To assess original studies for use in valuing estimated changes in well nitrate levels under revised CAFO regulations, EPA evaluated the *applicability* and the *quality* of the original studies on several criteria. To the extent feasible, EPA obtained or derived information from each of the

reports or papers for 28 categories of information used to characterize the studies. Because applicability to CAFOs and quality of the value estimates are distinct concepts, EPA evaluated these characteristics of the studies separately. Overall, the goal of the rating process was to identify studies that elicited high-quality value estimates (reliable and valid) and which were most applicable to the benefits assessment. There were three steps in the rating process:

- (1) identify study characteristics upon which to judge applicability and quality;
- (2) assign scores to the studies based on these characteristics;
- (3) assign weights to these scores for aggregating scores into unidimensional measures of applicability and quality.

Criteria for Ranking based on Applicability

Applicability refers to the relationship between values elicited in the primary groundwater valuation studies and benefit estimates necessary for application to the analysis of revised CAFO regulations. EPA's criteria for evaluation of applicability included comparison of the following characteristics of studies with likely CAFO situations:

- location (urban, rural, etc.);
- water supply/groundwater use (percentage on wells);
- type of contaminants (scenario involved nitrate contamination of groundwater);
- source of contaminants (scenario involved conditions similar to those relevant for CAFOs);
- value estimates are for the correct theoretical construct (e.g., total willingness to pay for reducing groundwater contamination from nitrates).

Criteria for Ranking based on Quality

Analysis of study quality was based on evaluation of the validity and reliability of the value estimates derived in the primary groundwater valuation research. Most of the 11 identified studies involved stated preference elicitation using survey methods. Based on professional experience as to what constitutes a valid and reliable stated preference valuation study, EPA identified characteristics of these studies that indicate reliability and validity. Criteria for evaluation of study quality included:

- whether the study was published or peer reviewed;
- whether the survey implementation met professional standards;
- how many respondents there were and what the response rate was;
- whether and how the groundwater baseline was characterized and what change was presented in the groundwater scenario;
- whether the credibility of scenario change was assessed;
- what valuation method was used and whether it was appropriate for eliciting the intended value measures;
- the type and duration of payment vehicle;
- whether appropriate empirical estimation was undertaken;
- whether expected explanatory variables were found to be significant.